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Faecal Sludge Drying: Can We Fake It?

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Faecal sludge drying: can we fake it?

Naomi Deering

A thesis submitted for the degree of Doctor of Philosophy

University of Bath

Department of Architecture and Civil Engineering

March 2020

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"It was the best of times, it was the worst of times..."

(Dickens, 2008)

To those who made it the best of times and who helped me through the worst.

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Abstract

The new emphasis on safely managed sanitation brought about by the Sustainable Development Goals era has placed faecal sludge management (FSM) firmly at the centre of the development agenda. FSM refers to everything in the sanitation after containment i.e. the removal, transport and reuse or disposal of faecal sludge (FS). A failure at any point in the FSM stream can have significant impacts on public and environmental health due to exposure to untreated human waste. The causes of failure at each stage in the FSM stream are often associated with coast, location (of the sanitation facility or the treatment/disposal site) and a lack of understanding regarding the risks involved in emptying untreated faecal sludge (FS). Many countries attempt to solve the FSM problem with large-scale, centralised wastewater and sewerage systems. However, these solutions are often unsuitable and unsustainable in the context of developing countries due to construction, operation and maintenance costs. Instead, a stepwise, context-specific approach to FSM is required.

The focus of this thesis is the treatment of FS on unplanted sludge drying beds (USDBs) with a filter. USDBs are simple, low-cost FS treatment options for rural and peri-urban areas. As such it is desirable to improve their design and efficiency. Two primary mechanisms are responsible for the dewatering of FS on USDBs: filtration and evaporation. Several factors affect the efficiency of FS treatment on USDBs including climatic conditions, the operating parameters of the bed itself (sludge loading rate and thickness and quality of the filter media) and the characteristics of the influent sludge. The type of sludge plays a key role in whether it can be treated on an unplanted drying bed. It has been found that stabilised sludges dewater well while fresh sludge is often unsuitable for treatment on USDBs due to its resistance to settling. As such, it requires mixing with more stabilised FS prior to being deposited onto the drying bed.

It has been observed in practice that up to 80% of influent FS volume deposited on a drying bed will emerge as effluent. It is common for this to occur within the first 24 hr – 48 hr. This effluent requires treatment prior to discharge into the environment. It is desirable to be able to predict the likely volume of effluent runoff based on the characteristics of the influent FS in order to prevent the over or under designing of the effluent treatment technology. Additionally, it is often the case that sludge dryness is estimated by operators based on observations; however, this can result in premature or late removal of the sludge solids. Both scenarios can cause problems. Premature removal of sludge solids may affect the efficiency of further treatment while late removal of the biosolids may result in the USDB not being

available for new influent sludge which may then be disposed of untreated into the environment.

It would be useful for practitioners to combine practical observations with a model based on external conditions and influent sludge properties capable of predicting the necessary sludge drying time required to achieve a particular moisture content. Such a model should then prevent the under or over designing of treatment facilities as well as ensuring efficient secondary treatment following dewatering. To achieve this goal an in-depth understanding of the FS properties governing its dewatering behaviour is necessary. However, FS is highly variable and concentrated, as such it is difficult to perform repeatable tests to assess dewatering behaviour and the properties affecting it. Therefore, the purpose of this thesis was the development of a synthetic FS simulant which would replicate the key properties affecting dewatering of stabilised FS deposited on an unplanted drying bed. This simulant could then be used to perform safe and repeatable drying tests under varying conditions to develop a useful model for the prediction of drying time required to achieve a given moisture content.

To date, no simulant exists which replicates the key properties affecting the dewatering and drying of real, stabilised FS. A comprehensive literature review was conducted encompassing the well-developed wastewater sludge dewatering theory. A list of key properties found to affect the dewatering of stabilised FS in terms of filtration and evaporation was collated. These properties were used in the development of eleven new FS simulants based on two simulants developed by PRG (2014) and Penn *et al* (2018). The simulants were characterised, and drying tests were conducted to assess the representativeness of the simulants in terms of the key properties and drying behaviours.

Results showed that the simulants performed marginally better than previous simulants in terms of replicating the properties of stabilised FS. Shrinkage and crust formation were observed during the drying tests, but no cracking as would be expected for real FS. A successful simulant was not found but several key observations were made. A research gap was identified in terms of a lack of research regarding the properties of sludge which affect moisture loss by evaporation. This is significant as solar drying and thermal drying of FS are promising new treatment options. Therefore, research into the properties of influent sludge which will determine its drying rate will aid the efficiency of these technologies. Furthermore, to the best of the author's knowledge, this is the first thesis to provide a comprehensive list of the properties which are likely to affect the drying of FS by filtration and evaporation. This will assist in future research regarding simulant development to replicate the key properties affecting the dewatering of sludge

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Nomenclature

a, b, c, x_0, y_0	Empirical constants
A_{sl+con}	Mass of wet sludge and container
B_{s+con}	Mass of dry sludge and container
c	Specific heat capacity
C	Volumetric heat capacity
C_{con}	Mass of container
Cp_s	Specific heat capacity of dry sludge
Cp_{sl}	Specific heat capacity of wet sludge
Cp_w	Specific heat capacity of water
d_{10}	Grain sizes 10% smaller by volume
d_{90}	Grain sizes 90% smaller by volume
D_{s+cru}	Mass of dry sludge and crucible before ignition
DR	Drying rate
e	Sample thickness
E	Dimensionless process time
E_{s+cru}	Mass of dry sludge and crucible after ignition
erf(E)	Error function
F_{cru}	Mass of crucible
k, k_1, k_2	Drying constant
k	Permeability
M	Sample mass at any instant
M_0	Initial mass
M_e	Mass at equilibrium moisture content
MR_M	Mass ratio
M_{pan}	Mass of pan
$M_{pan+sample}$	Mass of pan and sample
M_{sample}	Mass of sample
M_s	Mass of solids
M_w	Mass of water
MR_X	Moisture ratio
n	Drying exponent
N	Rotational speed
N_0	Initial drying rate
P	Pressure

R^2	Coefficient of determination
R_c	Centrifuge arm length
S_u	Undrained shear strength
t	Time
t_{2s}	Two stage drying time
t_{cr}	Constant rate drying time
t_f	Final drying time
T	Temperature
T_a	Air temperature
T_0	Temperature at centre
V	Sample volume
w	Moisture content
w_l	Liquid limit
w_p	Plastic limit
W_{c1}	Moisture content at first point of inflection of Kirscher plot i.e. critical moisture content
W_{c2}	Moisture content at second point of inflection of Kircher plot
W_{eq}	Equilibrium moisture content
X	Moisture content
X_0	Initial moisture content
X_d	Water removed by the dewatering process
X_e	Equilibrium moisture content
X_f	Unbound water
X_F	Theoretical best performance to be expected of any dewatering device
X_i	Initial moisture content

α	Thermal diffusivity
δ_1, δ_2	Interval of temperature variations
γ	Shear rate
λ	Thermal conductivity
μ	Viscosity
σ	Characteristic time
ρ	Density
ρ_b	Bulk density
ρ_d	Apparent (dry) density
$\rho_{particle}$	Particle density

ρ_t	Total density
ρ_w	Density of water
τ	Shear stress
χ^2	Reduced chi square
ψ_c	Pore water pressure
ψ_o	Osmotic pressure

Abbreviations

ABR	Anaerobic Baffled Reactor
APHA	American Public Health Association
ASTM	American Society for Testing and Materials
BBS	Bangladesh Bureau of Statistics
BCG	Boston Consulting Group
BH	Buro Happold Engineering
BM	Bath mix
BMP	Biochemical methane potential
BOD_5	Biochemical oxygen demand
Bott	Bottom
BS	British Standards
BSI	British Standards Institution
BST	Bristol Stool Chart
CAB	Community ablution block
CIAT	International Centre for Tropical Agriculture
C (TC)	Carbon (total carbon)
Ca	Calcium
Cl	Clay
CH_4	Methane
CHV	Calorific heat value
$C_{inorganic}$	Inorganic carbon
Cl	Chlorine
Cl^-	Chloride
Cml ST	Commercial septic tank
Co	Compost
CO	Carbon monoxide
CO_2	Carbon dioxide
COD	Chemical oxygen demand
COD_b	Biodegradable chemical oxygen demand
COD_{col}	Colloidal chemical oxygen demand
COD_{dis}	Dissolved chemical oxygen demand
COD_f	Filtered chemical oxygen demand
COD_{fats}	Chemical oxygen demand of fats present
COD_p	Particulate chemical oxygen demand

COD_{pb}	Particulate biodegradable chemical oxygen demand
COD_{poly}	Chemical oxygen demand of polysaccharides present
COD_{prot}	Chemical oxygen demand of proteins present
COD_{pu}	Particulate unbiodegradable chemical oxygen demand
COD_{s}	Soluble chemical oxygen demand
COD_{sb}	Soluble biodegradable chemical oxygen demand
COD_{ss}	Suspended chemical oxygen demand
COD_{su}	Soluble chemical unbiodegradable oxygen demand
COD_{u}	Unbiodegradable chemical oxygen demand
CP	Cesspool
CST	Capillary suction time
DAF	Dry ash free
DO	Dissolved oxygen
DS (TDS)	Dissolved solids (total dissolved solids)
DVB	Divalent Cation Bridging Theory
DVLO	Double Layer Theory
EC	Electrical conductivity
Ecosan	Ecological sanitation
EFD	Excreta flow diagram
EPS	Extracellular polymeric substances
FC	Fixed carbon
FF	Fresh faeces
Fl	Fluidised
FS	Faecal sludge
FSM	Faecal sludge management
GA	Gas analysis
GPR	Gas production rate
H	Hydrogen
HHV	Higher heating value
HTD	Hydrotextural diagram
ISO	International Organisation for Standardisation
IQR	Interquartile range
JMP	Joint Monitoring Programme
K (TK)	Potassium (total potassium)
LaDePa	Latrine dehydration and pasteurisation
LALLS	Low angle laser light scattering

Ma	Maize
m/c	Moisture content
MDG	Millennium Development Goals
Mg	Magnesium
N (TN)	Nitrogen (total nitrogen)
Na	Sodium
ND	Naomi Deering
N.D.	No data
NH_3 (-N)	Ammonia
NH_4^+ (-N)	Ammonium
NIPORT	National Institute of Population Research and Training
NO	Nitric oxide
NO_2^- (-N)	Nitrite
NO_3^- (-N)	Nitrate
N_{org}	Organic nitrogen
N/S	Not specified
NR	Not reported
O	Oxygen
OC ($C_{organic}$)	Organic carbon
ORP	Oxidation-reduction potential
OSS	Onsite sanitation system
P (TP)	Phosphorous (total phosphorous)
PEG	Polyethylene glycol
ppm	Parts per million
PO_4^{-3} (-P)	Orthophoshpate
PSC	Particle size composition
PSD	Particle size distribution
Q2	Median
RH	Relative humidity
RMSE	Root mean square error
S	Sulphur
SB	Simulant containing baker's yeast
Sch	School
SDG	Sustainable Development Goals
SE	Simulant containing only yeast extract
SG	Specific gravity

SLR	Sludge loading rate
SO_4^{-2}	Sulfate
SPR	Sludge penetration resistance
SRF	Specific rate of filtration
SS (TSS)	Suspended solids (total suspended solids)
SSE	Sum of squares due to error
SS9	Synthetic sludge 9
ST	Septic tank
SuSanA	Sustainable Sanitation Alliance
SVI	Sludge volume index
TKN	Total kjedhal nitrogen
TS (DM)	Total solids (dry matter)
UASB	Upflow Anaerobic Sludge Blanket Reactor
UDT	Urine diversion toilet
UDDT	Urine diversion dehydration (or dry) toilet
UK	United Kingdom
UN	United Nations
UNDG	United Nations Development Group
UNDP	United Nations Development Programme
UNICEF	United Nations International Children's Emergency Fund
USDB	Unplanted sludge drying bed
US EPA	United States Environmental Protection Agency
VFA	Volatile fatty acids
VS (TVS)	Volatile solids (total volatile solids)
VSS (TVSS)	Volatile suspended solids (total volatile suspended solids)
WASH	Water, sanitation and hygiene
WHO	World Health Organisation
WRC	Water retention curve
WSP	Water and Sanitation Program
WSSCC	Water Supply and Sanitation Collaborative Council

1 Introduction and Background

1.1 Background

In 2000 world leaders gathered at the United Nations (UN) headquarters in New York to commit themselves to "a new global partnership to reduce extreme poverty, and set out a series of eight time-bound targets – with a deadline of 2015 – that [became] known as the millennium development goals (MDGs)" (United Nations, 2016). According to the then General-Secretary of the UN, Kofi Annan, "unsafe water and poor sanitation cause an estimated 80[%] of all diseases in the developing world… No single measure would do more to reduce disease and save lives in the developing world than bringing safe water and adequate sanitation to all" (Annan, 2000). Acknowledging the vital importance of water and sanitation, as part of Goal Seven, the MDGs sought to "halve, by 2015, the proportion of the population without sustainable access to safe drinking water and basic sanitation" (United Nations, 2015a).

The indicator used to measure progress for sanitation coverage was the "proportion of the urban and rural population with access to improved sanitation" (UNDG, 2003). According to the WHO/UNICEF Joint Monitoring Programme for Water Supply, Sanitation and Hygiene (JMP) (2000), technology was used to provide an approximate description of sanitation coverage as 'improved' or 'unimproved'. The primary consideration in establishing whether a technology is improved or unimproved is whether it allows the disposal of human waste in a manner which does not contaminate the environment and limits the likelihood of transmission of disease from person to person (WSSCC, 2000). To facilitate analysis of national data a standard set of categories for sanitation i.e. a sanitation 'ladder', was established by the JMP (WHO and UNICEF JMP, 2014) as shown in Table 1-1. Between 1990 and 2015 2.1 billion people gained access to an improved sanitation facility; however, the target set in 2000 was ultimately missed and 2.3 billion people still lacked access to even a basic (i.e. not shared) sanitation service by the end of the MDG period (United Nations, 2015a; WHO and UNICEF JMP, 2017b).

Table 1-1. The JMP ladder for sanitation services during the Millennium Development Goals (MDG) period (adapted from WHO and UNICEF JMP 2014)

	MDG Sanitation Ladder	Definition
Improved Sanitation	Improved	Facilities which are likely to ensure hygienic separation of human excreta from human contact including: - flush or pour flush latrine connected to a piped sewer system, septic tank or latrine pit; - ventilated improved pit (VIP) latrine - pit latrine with slab or squatting plate - composting toilet
Unimproved Sanitation	Shared	Use of improved facilities shared between two or more households. Only facilities that are not shared or not public are considered improved.
	Unimproved	Facilities which do not ensure hygienic separation of human excreta from human contact such as pit latrines without a slab or platform, hanging latrines or bucket latrines.
	Open Defecation	Disposal of human faeces in fields, forests, bushes, open bodies of water, beaches or other open spaces, or with solid waste.

With the advent of the SDG era in 2015, there came a new emphasis on safely managed sanitation and the JMP introduced an updated sanitation ladder as shown in Table 1-2 Improved sanitation facilities were split into three categories: basic; limited (i.e. shared); and safely managed. The addition of limited facilities to qualify as improved was primarily because it was acknowledged that, due to cultural practices and socioeconomic constraints in densely populated areas, high-quality shared facilities may be the most appropriate short-term option in some contexts (WHO and UNICEF JMP, 2017b). However, the new ladder serves to highlight safely managed sanitation as the ultimate goal. Approximately 39% of the global population has access to a safely managed sanitation service (WHO and UNICEF JMP, 2017b).

Table 1-2. The new JMP ladder for sanitation services during the Sustainable Development Goals (SDG) period where improved facilities include flush or pour/flush toilets connected to septic tanks; protected latrines such as VIP latrines or pit latrines with slabs/squatting plates; and composting toilets (adapted from WHO and UNICEF JMP 2017b).

	SDG Sanitation Ladder	Definition
Improved Sanitation	Safely Managed	Use of improved facilities that are not shared with other households and where excreta are safely disposed of in situ or transported and treated offsite.
	Basic	Use of improved facilities that are not shared with other households.
	Limited	Use of improved facilities shared between two or more households.
Unimproved Sanitation	Unimproved	Use of pit latrines without a slab or platform, hanging latrines or bucket latrines.
	Open Defecation	Disposal of human faeces in fields, forests, bushes, open bodies of water, beaches or other open spaces, or with solid waste.

To have a safely managed sanitation service, human waste must be safely disposed of either in situ or transported and treated offsite. To achieve this, improved sanitation facilities must be connected to either a sewer network i.e. a sewered facility, or to an onsite sanitation system (OSS) such as a latrine pit or septic tank[1] i.e. a non-sewered facility (WHO and UNICEF JMP, 2017b). It is necessary to distinguish between sewered and non-sewered facilities as they require different approaches for excreta management. In a sewered facility, excreta are flushed and transported away from the household through piped sewerage to a wastewater treatment plant. The combination of excreta, blackwater and cleansing material (plus greywater) is commonly referred to as domestic wastewater. Excreta which originates from an OSS and has not been transported through a sewer is described as faecal sludge (FS) (Strande et al., 2014). FS may be "raw or partially digested, a slurry or semisolid, and results from the collection, storage or treatment of combinations of excreta and blackwater, with or without greywater" (Strande et al., 2014). FS is either emptied, transported and treated off-site or treated and disposed of in situ. This is a significant assumption, as OSS may be left to overflow or leak, either deliberately or accidentally, and therefore the sludge is not safely managed but instead enters the environment untreated. Furthermore, where an OSS is

[1] As noted in WHO and UNICEF (2017b) many onsite storage systems should be classified as simple vaults or cesspools given that they often lack the defining design features of a septic tank i.e. watertight walls and floor, multiple chambers separated by baffles and an outlet pipe leading to a soak pit or leachfield. However, the term 'septic tank' is commonly used in literature to cover many kinds of onsite systems and that is how it is used in this thesis.

emptied it may not qualify as a safely managed sanitation service either. Poor understanding of the risks involved in both emptying and disposing of the sludge means that often the untreated FS is simply dumped in a nearby field, water body or open drain (Ingallinella et al., 2002). It has been estimated that up to 90% of all wastewater in developing countries is discharged untreated directly into rivers, lakes or the oceans (United Nations, 2015b).

According to the Boston Consulting Group (BCG) (Cairns-Smith et al., 2014) there are currently 2.7 billion people reliant on OSS to manage their waste with this number predicted to increase to 4.9 billion if current trends in sanitation continue. Thus, faecal sludge management (FSM) is of key importance. FSM refers to everything in the sanitation chain after containment i.e. the removal, transport, treatment and reuse or disposal of faecal sludge (FS), as shown in Figure 1-1.

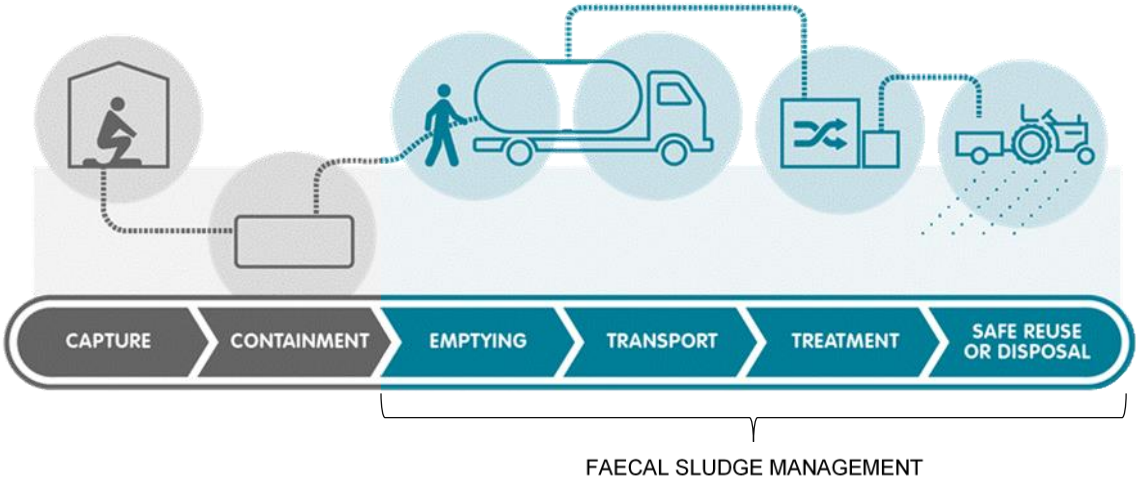


Figure 1-1.The sanitation chain (SuSanA, 2017)

Without the subsequent success of each step in the FSM 'stream', people and the environment are at risk of exposure to untreated human waste. This undermines the gains made through increased sanitation coverage which has significant social and economic impacts. Untreated waste may leak into groundwater which is often used as a source of drinking water or it may end up in water used by downstream communities resulting in exposure to pathogens (de Albuquerque, 2010). Open defecation and the indiscriminate disposal of untreated FS are key factors in the spread of diseases such as cholera, diarrhoea and dysentery as well as intestinal worms, schistosomiasis, and trachoma (WHO, 2015; World Bank, 2016). Diarrhoea alone results in the deaths of more than 750,000 children under five every year while 272 million school days per year are lost to the illness (UNU-INWEH, 2015). It is also estimated that every 1USD invested in better sanitation delivers up to 5USD[2] in social

[2] Other estimates have set this figure at 9USD (Cairns-Smith et al., 2014).

and economic benefits through increased productivity, reduced healthcare costs and the prevention of illness, disability and early death (UNU-INWEH, 2015).

FSM is critical to human health and wellbeing where OSS are the dominant form of sanitation system; however, the manner in which a safely managed sanitation service is achieved is a complex problem. As shown in Table 1-3 the causes of failure at each stage in the FSM stream are multiple and are often associated with cost, location (either of the OSS facility or the treatment/disposal site) and lack of understanding regarding the risks involved in emptying and/or disposing untreated FS. In many countries political and administrative preferences tend towards large-scale, centralised wastewater and sewerage systems (Arbogast and Jaehyang, 2013). However, such solutions are often unsuitable and unsustainable in the context of developing countries given the high construction, operation and maintenance costs (Koné and Peter, 2008; Cairns-Smith et al., 2014). Additionally, according to Strande *et al* (2014) it is often the case that, "where sewers and wastewater treatment plants have been constructed in low-income countries, they have… resulted in failures." To address the issues highlighted in

Table 1-3 requires a tailored and context specific approach accounting for diverse local conditions and resource constraints (de Albuquerque, 2010). A broad portfolio of solutions is required with "an increasing focus on more flexible decentralised and on-site systems likely" (Cairns-Smith et al., 2014).

Table 1-3. Current faecal sludge management practices - causes and consequences (adapted from (Strauss and Montangero, 2002)*)*

FSM Component	Causes	Consequences
Emptying	• Limited/no accessibility to pits/septic tanks • Inappropriate emptying equipment • Manual, non-mechanical emptying	• Overflowing pits/septic tanks • Low emptying frequency
Transport	• Lack of suitable disposal or treatment sites at a short distance from area of FS collection • Traffic congestion	• Indiscriminate disposal of FS at shortest possible distance from point of collection
Treatment	• Emptiers don't want to pay to dispose FS at official treatment/disposal sites	• FS is used or disposed of untreated
Reuse	• Lack of awareness of risks in reuse of untreated FS • Lack of willingness to pay treatment fees	• Environmental pollution from use of untreated FS in agriculture and aquaculture
Disposal	• Lack of suitably close treatment sites • Lack of treatment sites altogether • Lack of incentives for disposal at treatment sites • Lack of awareness of risks posed due to indiscriminate disposal of FS	• Indiscriminate disposal of FS

Given the previous emphasis on capture and containment of excreta during the MDG era, there is a significant likelihood that some form sanitation technology may already be in place in a particular area. At least in the short term, FSM solutions should work with these existing technologies to ensure safely managed sanitation (United Nations, 2015a). To develop suitable FSM solutions a stepwise and context specific approach is necessary, and each stage of the FSM stream may be addressed separately. The technology required for emptying is often dependent on accessibility as well as the capture and containment technology in use. The type of capture and containment technology also affects treatment and reuse/disposal options as it has a significant impact on the type of FS produced (Heinss et al., 1998; Strande et al., 2014). Transport options for sludge are often limited by the geographical context i.e. urban versus rural. It is often the case that suitable treatment sites for FS are found on the outskirts of urban areas meaning transport distances are large rendering the transport of relatively small volumes of through congestion and over long distances unsustainable (Montangero and Strauss, 2004). This is even more of a pertinent issue where manual

emptying, either by hand using buckets and shovels or portable, manually operated pumps such as vacutugs, is required (WaterAid, 2018).

The focus of this thesis is the treatment of FS by drying on covered, unplanted sludge drying beds (USDBs). It is the result in coordination with a request made to Buro Happold Engineering (BH) Bath from WaterAid Bangladesh to recommend a suitable solution for the treatment of FS from various OSS in Bangladesh. The BH Bath pilot project served to provide the focus of drying FS on a USDB considered in this thesis as well as to provide the basis for the meteorological parameters used for drying tests. The lead engineer on the pilot project in Bangladesh, Dr Celia Way, was also an external supervisor for this work but there was no direct involvement of BH Bath over the course of this project. Appendix A provides a detailed outline of the pilot project run by BH Bath and the rationale behind the selection of covered, USDBs for FS treatment in Bangladesh. The operation and primary factors affecting the performance of a USDB are discussed in the following section.

1.2 Unplanted Sludge Drying Beds

The main objective of FS treatment on USDBs is solid-liquid separation, or dewatering. Two primary mechanisms are responsible for FS dewatering on USDBs with a filter bed: filtration and evaporation, as shown in Figure 1-2. Solids are trapped on the surface of the filter bed, while liquid either percolates through the filter bed and is then collected in a drain, or evaporates from the surface of the sludge (Strande et al., 2014). Although studies (Cofie et al., 2006; Koné et al., 2007) have found that the majority of total solids and pathogens are contained within the solids on top of the filter, the effluent still requires treatment prior to discharge into the environment due to potentially high ammonia and salinity. Pond systems, storage tanks, soak pits and planted drying beds in series may all be employed to treat the effluent from a drying bed depending on its characteristics and intended end use (see Appendix A for more details) (Koné and Strauss, 2004; Kengne et al., 2011; Kengne et al., 2013). Once the biosolids has reached the desired reduction in moisture content it must be removed either manually or mechanically from the drying bed. The removed biosolids then require further processing to allow increased stabilisation and pathogen removal, the thoroughness of which will depend upon the intended end use.

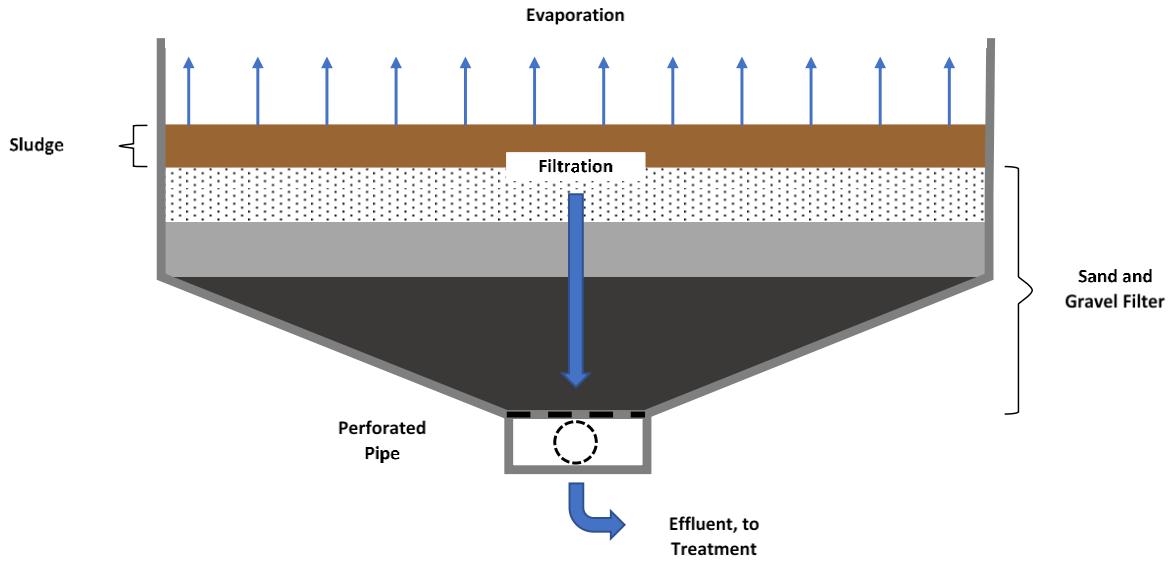


Figure 1-2. Unplanted drying bed profile (adapted from Dodane & Ronteltap 2014)

Based on practical observations and field experiments, the performance of an unplanted drying bed is dependent on climatic conditions, the operating parameters of the drying bed itself and the characteristics of the influent FS (Heinss et al., 1998). Ambient weather conditions will affect both the rate of filtration and evaporation from the drying beds. For evaporation to occur from a body there must be enough energy supplied to it to meet the latent heat of vaporisation[3] requirement and both a pressure gradient between the body and the atmosphere and a mechanism for vapour removal must exist (Hillel, 1998). Therefore, high humidity and low temperatures and windspeed will reduce the rate of evaporation while high temperatures in combination with relatively low humidity and a high wind speed will enhance the total amount of water removed via evaporation (Dodane and Ronteltap, 2014). The rate of filtration of FS deposited on an uncovered drying bed has been found to be prolonged by rainfall depending on the level of dryness of the biosolids (Cofie et al., 2006; Pescod, 1971). Where biosolids were relatively dry and cracking had occurred, exposing the filter media, rainwater drained quickly through the cracks without increasing the moisture content in the biosolids (Pescod, 1971). If free moisture from the sludge had not drained, it was observed that rainfall increased the moisture content of the sludge, extending the rate of filtration.

Operational and design factors affecting the performance of drying beds include sludge loading rate (SLR), sludge thickness and the quality of the filter media. SLR represents "the mass of solids dried on one m^2 of bed in one year" (Dodane and Ronteltap, 2014). High SLRs can result in low rates of evaporation as the greater the total mass of an object the greater the

[3] Latent heat of vaporisation is the energy needed to break hydrogen bonds.

amount of energy that can be stored and consequently the greater the amount of energy necessary to meet the latent heat of vaporisation requirement (Strande et al., 2014). Similarly, the thickness of the sludge layer will impact the rate of evaporation from the sludge surface due to the increase or decrease in total mass. Furthermore, for evaporation to be continuous there must be "a continual supply of water from or through the interior of the body to the site of evaporation" (Penman, 1948; Hillel, 1998). Therefore, if sludge is applied in a thick layer, there is further for moisture to be transported to the evaporating surface as the sludge dries. The preference is for the FS to be applied in a layer of 20cm to 30cm in depth. Pescod (1971) found that maximum allowable SLR can be achieved with a sludge application depth of 20 cm after conducting experiments with FS of variable total solids (1.7% to 6.5%) and loading depths during both dry and rainy season. Cofie and Koné (2009) used a sludge load depth of 30 cm. The experiments conducted by the Ghana Water Research Institute in Accra also used a sludge loading depth of 20 cm and the loading depth for the pilot study in Bangladesh was between 20 cm and 30 cm (Heinss et al., 1998; Way, 2013a).

The quality of the filter media affects the rate of filtration on a drying bed (Strande et al., 2014). Cofie *et al* (2006) found that the sand particles of the filter crumbled and clogged over time reducing rate of infiltration and, consequently, rate of filtration. Kuffour *et al* (2009) investigated the appropriate particle sizes of sand to improve FS dewatering efficiency of USDBs to enhance treatment of large volumes of FS. Three configurations of filter were used in the research. Coarse and fine gravel of 2.0cm to 3.0cm and 0.5cm to 1.0cm in diameter respectively were placed into each drying bed to 15cm depth and 10cm depth respectively. The sand was placed to a depth of 20cm and was either 0.1mm to 0.5mm, 0.5mm to 1.0mm or 1.0mm to 1.5mm in diameter. The depths were within the recommended limits for design of a USDB (Strande et al., 2014). Kuffour *et al* (2009) concluded that the bed with the smallest sand particle size was most efficient in terms of contaminant load removal whilst those with larger sand particle sizes were likely to clog faster.

The type of FS is a key factor in the efficiency of USDB performance. The rate and extent of filtration of untreated faecal sludges are likely to be much lower than that of stabilised sludges (Strauss et al., 1997). Heinss *et al* (1998) separated FS into two broad categories based on origin and degree of stabilisation. Type A sludge is mostly fresh and originate from non-flush or pour flush toilets and bucket toilets which require more frequent emptying (Heinss et al., 1998). Type B sludge originates from septic tanks or pits where it is stored for a year or more and undergoes biochemical stabilisation to a considerable extent (Heinss et al., 1998). Type B sludge is more suitable for drying on USDBs as it settles more easily. Type A sludge may also be dried on drying beds but requires mixing with type B sludge prior to being deposited

on the bed (Heinss et al., 1998; Koné and Peter, 2008). Results from a study by the Ghana Water Research Institute in Accra found that mixtures of type A to type B sludge at a ratio of 1:4 exhibited good filterability, drying to a maximum of 70% total solids in eight days (Heinss et al., 1998). Koné and Strauss (2004) recommended a lower ratio of 1:2-3 type A sludge to type B sludge and Cofie *et al* (2006) used a ratio of 1:2 type A sludge to type B sludge. Cofie *et al* (2006) found that drying time to achieve a total solids content of 20% on a USDB with sand and gravel filter varied from one week to two months, with an average of two weeks. Using a ratio of 1:2 type A: type B sludge, Cofie and Koné (2009) found that sludge could be removed from the drying beds after 10 days, similar to the findings of the BH Bath pilot study (Way, 2013a).

The type of water in FS is also a key factor in determining both the rate of filtration and evaporation. The water in sludge may be spilt into free water and bound water, where free water is water that is easily removed by mechanical means while bound water is only removable thermally. In other words, free water is the filtrate from the sludge deposited on the drying bed and bound water is the water which is evaporated from the surface of the sludge. It has been observed in practice that anywhere between 50.0% and 80.0% of the FS volume discharged onto a drying bed will emerge as drained liquid normally within the first 24 – 48 hours (Koné and Peter, 2008). Consequently, evaporation will be responsible for between 50.0% and 20.0% of moisture loss (Heinss et al., 1998). The proportions of free water and bound water vary depending on the sludge characteristics and will impact which process, filtration or evaporation, is the dominant dewatering mechanism. For thickened sludge the filtration rate has been found to be very low and the volume of effluent was considered negligible in comparison to the incoming volume of sludge (Badji et al., 2011). Badji *et al* (2011) reported an effluent volume of 0.516 m^3 out of the 55.0 m^3 of sludge deposited onto drying beds after four days. In this case, the primary drying mechanism was evaporation rather than filtration. This is in keeping with findings from Pescod (1971) who reported that septage sludge with a higher initial total solids content took longer to drain and therefore evaporation became the dominant drying mechanism.

Despite the high land requirement of USDBs they remain a key, low-cost, low maintenance FS treatment option for rural and peri-urban areas and it is important to improve their design and efficacy. It is often the case that the level of dryness of sludge is estimated by operators based on observation for example, by the amount of cracking of the biosolids. However, this can often result in biosolids being removed too soon or being left on the drying beds longer than necessary (Badji et al., 2011). Both scenarios are problematic. Premature removal of biosolids may impact further treatment. For example, the optimum moisture content for co-

composting is 40.0% - 60.0% by weight, a higher moisture content can result in the development of anaerobic conditions which can impede microbial growth (Strande et al., 2014). Where biosolids are left on the drying beds longer than necessary, this can affect the volume of sludge which can be treated. This may have significant health and environmental consequences should the surplus sludge simply be disposed of untreated. Alongside observational judgements, a model to predict the required drying time to achieve a required moisture content (determined by the intended end-use) would be useful for practitioners to ensure sufficient dewatering of sludge prior to removal and treatment and prevent over or under-designing of treatment facilities. Furthermore, given the potentially significant volume of filtrate that can drain in a short time when influent FS is first deposited onto an unplanted drying bed it is vital to ensure that any leachate treatment is designed to accommodate it. Therefore, predictions of the filterability of influent sludge should prevent over or under-designing of leachate treatment facilities.

To achieve these goals, it is necessary to develop an in-depth understanding of the properties of FS which govern its dewatering behaviour. However, there is a current lack of detailed information regarding the key characteristics affecting FS dewatering. As FS is highly variable it has resulted in significant difficulty in performing controlled analyses of the factors that determine its dewatering behaviour (Ward et al., 2017). Additionally, handling FS is a serious health and safety risk due to the high pathogen content. To address these issues, a number of synthetic FS simulants which are representative of the properties of real FS dried on USDBs i.e. stabilised sludge, were developed.

1.3 Justification for Research

The following justifications for this research can be made:

- At present, to the author's knowledge, there is no FS simulant which replicates the key properties affecting the dewatering behaviour of real stabilised FS. Such a simulant will allow the prediction of sludge retention times to achieve required moisture contents under given climatic conditions as well as aiding in preventing the over- or under-design of both drying beds and leachate treatment technologies. This will ensure more efficient use of USDBs and available resources.

- A successful FS simulant is useful in the wider context of FSM as a research tool for the development of new FS drying technologies as it provides a substance which should mimic the behaviour of real FS whilst being safe to handle and enables controlled analysis as it is a homogenous material. This will ensure continuing

improvements in FS treatment and ultimately ensure the safety of the environment and public health by reducing the volume of FS disposed of untreated.

1.4 Research Questions

The following research questions are answered in this thesis:

R1. What are the key properties which affect the dewatering behaviour of FS?

R2. Is it possible to replicate these properties is a synthetic simulant?

R3. Will a synthetic simulant replicate the drying behaviour of real FS deposited onto USDBs?

R4. Can a model be developed to predict the necessary sludge retention time to achieve a particular moisture content under particular climate conditions?

1.5 Research Objectives

Therefore, the research objectives are:

1. Provide a list of key properties which affect the dewatering behaviour of real FS deposited on USDBs.
2. Develop a number of simulants to represent the key properties highlighted in the literature review.
3. Compare the key properties of the developed FS simulants with results for the same properties of real FS to determine the most representative simulant.
4. Perform small-scale drying tests under controlled conditions to assess the drying behaviour of the FS simulants in comparison with that expected for real FS dried on an unplanted drying bed.
5. Perform small-scale drying bed tests to assess the rate of filtration and evaporation of synthetic simulant under different climate conditions.

Table 1-4 shows which research questions correspond to each of the objectives as well as the expected research outputs.

Table 1-4. Research outputs mapped against research questions and objectives

Chapter	Research Questions		Objectives		Research Outputs
2	R1.	What are the key properties which affect the dewatering behaviour of FS?	O1.	Provide a list of key properties which affect the dewatering behaviour of real FS deposited on USDBs.	List of key properties from literature review
3 4	R2.	Is it possible to replicate these properties is a synthetic simulant?	O2.	Develop a number of simulants to represent the key properties highlighted in the literature review.	Developed simulants
			O3.	Compare the key properties of the developed FS simulants with results for the same properties of real FS to determine the most representative simulant.	Experimental results and comparison of properties with real FS
5 6	R3.	Will a synthetic simulant replicate the drying behaviour of real FS deposited onto USDBs?	O4	Perform small-scale drying tests under controlled conditions to assess the drying behaviour of the FS simulants in comparison with that expected for real FS dried on an unplanted drying bed.	Experimental results
6 7	R4.	Can a model be developed to predict the necessary sludge retention time to achieve a particular moisture content under particular climate conditions?	O5.	Perform small-scale drying bed tests to assess the rate of filtration and evaporation of synthetic simulant under different climate conditions.	Results of application of models to experimental data

1.6 Key Definitions

The following key terms used throughout this thesis are defined as shown in Table 1-5.

Table 1-5. Key terms and definitions used in this thesis

Key Term	Definition
Faecal Sludge	Excreta which originates from an OSS and has not been transported through a sewer. It may be "raw or partially digested, a slurry or semisolid, and results from the collection, storage or treatment of combinations of excreta and blackwater, with or without greywater" (Strande et al., 2014).
Type A Faecal Sludge	Mostly fresh sludge which originates from non-flush or pour flush toilets and bucket toilets which require more frequent emptying (Heinss et al., 1998).
Type B Faecal Sludge	Sludge which originates from septic tanks or pits where it is stored for a year or more and undergoes biochemical stabilisation to a considerable extent (Heinss et al., 1998).
Dewatering	Taken to refer to any process by which FS loses moisture.
Filtration	A mechanism for dewatering; taken to refer to the loss of free water from FS through filter media on USDBs. May also be referred to as drainage.
Rate of filtration	The rate at which free water is lost from FS through filter media on USDBS.
Extent of Filtration	The solids remaining on the filter bed following filtration.
Evaporation	A mechanism of dewatering; defined as "the process in which a liquid changes to a gaseous state at the free surface below the boiling point through the transfer of heat energy" (Karamouz et al., 2012). May also be referred to as drying.
Biosolids	Taken to refer to the dewatered sludge left on the filter media of the drying bed (Cofie et al., 2006).

1.7 Thesis Outline

The remainder of this thesis is organised as follows:

2. **Literature Review**. A discussion of the theory relating to the dewatering behaviour wastewater sludge and FS. Key properties affecting dewatering behaviour are presented to inform the selection and development of a suitable FS simulant. Current FS simulants are discussed.

3. **Faecal Sludge Simulant Development**. Development of FS simulant to mimic the key properties outlined in the literature review. Methodology followed for mixing of the simulants is included and initial comparisons with original simulants are made to inform further development.

4. **Faecal Sludge Simulant Characterisation**. Methodology, results and analysis and discussion of the characterisation of the developed FS simulants.

5. **Preliminary Faecal Sludge Simulant Drying Tests.** Methodology, results and analysis and discussion of preliminary drying tests undertaken with climate chamber to monitor behaviour of developed simulants.

6. **Faecal Sludge Simulant Drying Tests.** Methodology, results and analysis and discussion of drying tests with UV light and a small scale USDB.

7. **Conclusions and Future Work.** A summary of the key findings of this work, limitations and recommendations for future work.

1.8 Summary

FSM has become one of the major challenges in the sanitation sector, both in Bangladesh and worldwide. Thanks to the SDGs, more emphasis has been given to safely managed sanitation instead of the previous focus on capture and containment. There is a great deal of work left to do if the SDG sanitation target is to be met by 2030 and more efficient, effective and sustainable solutions are required across the entire FSM stream. The focus of this thesis was FS treatment by drying on USDBs with a filter bed. The aim was to develop a synthetic FS simulant which would mimic the dewatering behaviour of real stabilised FS dried on an unplanted drying bed. This should allow safe, reproducible lab scale drying tests for the development of a model to predict the sludge retention time for a required moisture content under certain climate conditions.

2 Literature Review

This literature review is divided into five sections. In Section 2.1 , key properties which affect the filterability of wastewater sludge and FS are discussed. In Section 2.2 commonly observed behaviours exhibited by sludge during drying are discussed. The key properties which affect the drying rate of wastewater sludge and FS are outlined in Section 2.3 while Section 2.5 provides an overview of relevant sludge drying models. Finally, details of current research into FS simulants is presented in Section 2.5.

Although USDBs are a common low-cost FS treatment option, little research has been conducted into the properties which affect sludge dewatering (Ward et al., 2019). Instead most work to date has aimed to evaluate and improve design and operating conditions of drying beds rather than understand the role of sludge characteristics in dewatering (Vincent et al., 2012). Therefore, the scope of the literature review for Sections 2.1 and 2.2 was extended to include the well-established theory concerning the properties affecting the dewatering of wastewater sludges and observed drying behaviours. However, it is important to note that FS is markedly different to wastewater sludges (Ward et al., 2019). As such, not all properties will have the same effect on the dewatering behaviour of FS as they do on wastewater sludges.

2.1 Key Properties of Sludge Affecting Dewatering by Filtration

In this section, the key properties which affect dewatering of wastewater sludge and FS in terms of filtration are discussed.

2.1.1 Moisture Distribution

Extensive research has been conducted regarding the types of water found in wastewater sludge (Chen et al. 2002; Kopp and Dichtl 2000; Kopp and Dichtl 2001b; Vaxelaire and Cezac 2004; Vesilind 1994; Smith and Vesilind 1995; Vesilind and Hsu 1997). Initially the water in sludges was split into two types: unbound free water, which is removable by mechanical means, and bound water, which is only removable thermally. However, more recent literature suggests that there are four distinct types of water present in wastewater sludges based upon the type and intensity of their bonding to the solid sludge particles (Kopp and Dichtl 2000). The four types of water are outlined in Table 2-1 and their distribution in wastewater sludge is shown in Figure 2-1. No literature was found specifically regarding the distribution of water in FS. However, in an unplanted drying bed 50.0% - 80.0% of the FS volume applied has been observed to emerge as drained liquid (Heinss et al., 1998). Thus, there is likely to be a significant proportion of free, unbound water present in FS deposited onto USDBs depending

on the sludge properties. It also highly likely that FS contains interstitial water, vicinal water and intracellular water.

Table 2-1. Types of water present in wastewater sludge

Type of Water	Definition
Free unbound	Water which is not associated with or influenced by suspended solids particles in the sludge (Vesilind, 1994), it moves freely between individual sludge particles and is not influenced by capillary forces (Kopp and Dichtl, 2000). This type of water can be separated through mechanical means.
Interstitial (or immobilised)	Water which is trapped in the crevices and interstitial spaces of the flocs and organisms in the sludge and may be bound by active capillary forces between flocs (Kopp and Dichtl, 2000; Kopp and Dichtl, 2001b; Tsang and Vesilind, 1990; Vesilind, 1994).
Surface (or vicinal)[a]	Multiple layers of water molecules which are held tightly to particle surfaces by adsorptive and adhesive forces (Kopp and Dichtl, 2000; Kopp and Dichtl, 2001b; Tsang and Vesilind, 1990; Vesilind, 1994).
Intracellular[a]	Thought to contain water in cells and water of hydration, it is the water in sludge which is chemically bound to the sludge particles and only removable by the expenditure of thermal energy (Vesilind, 1994).

[a] Intracellular and surface water together are thought to make up the bound water content.

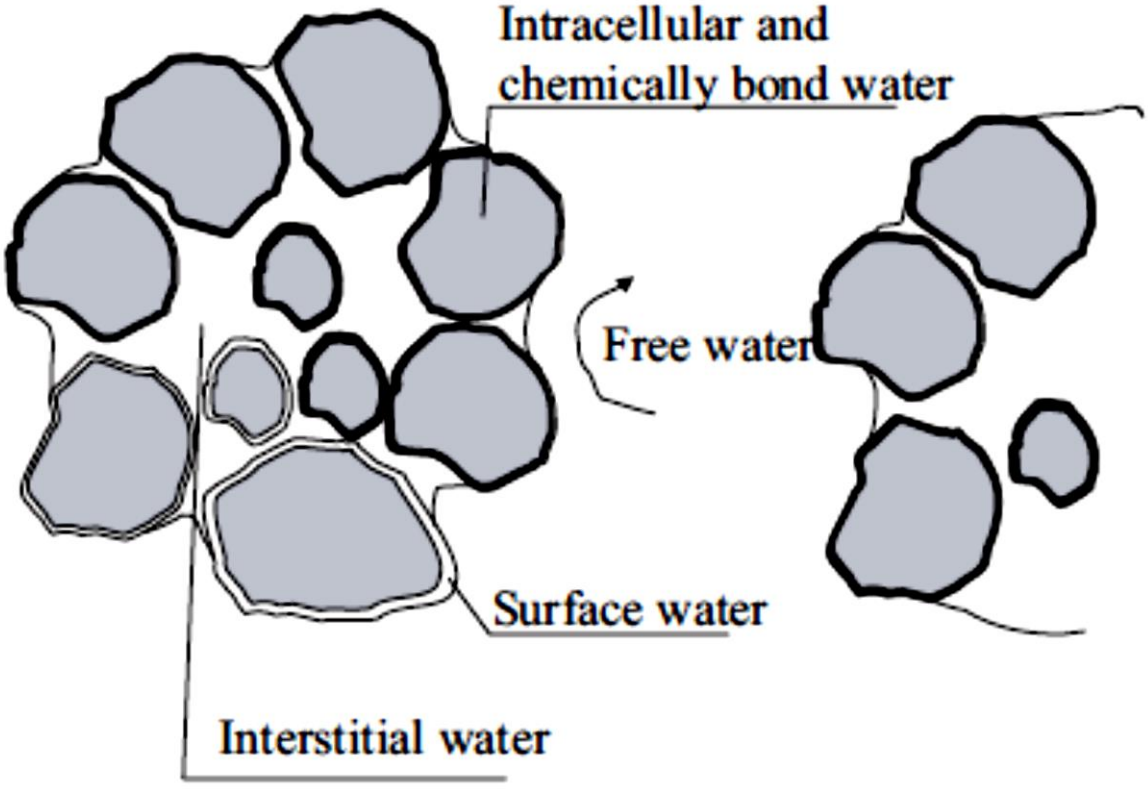


Figure 2-1. Moisture distribution in wastewater sludge (Chen et al., 2002)

The distribution of water in sludge, specifically the proportion of bound water, has been observed to significantly influence both rate and extent of filtration. The greater the volume of bound water, the less water available to drain freely and the more water retained in the dewatered sludge cake and vice versa. In wastewater theory, the representation of moisture distribution within sludge has long been considered essential for the examination of dewatering problems (Vaxelaire and Cezac, 2004). If the moisture distribution of a sludge can be successfully estimated, then predictions on the performance of dewatering processes can be made. For example, according to Tsang and Vesilind (1990) if mechanical dewatering devices (e.g. centrifugation or filter press) are only expected to remove unbound water, then the absolute maximum amount of water removable by a dewatering device can be represented by:

$$X_F = (X_d + X_f), \tag{2.1}$$

where X_F is the theoretical best performance to be expected of any dewatering devices, X_d is the water removed by the dewatering process and X_f is unbound water.

Thus, the distribution of water in FS is desirable as it may be used to predict both the rate and extent of filterability which can aid the design of dewatering technologies including USDBs. Methods for the determination of bound water content include: vacuum drying (Smollen, 1990); drying at atmospheric pressure (Chen et al., 2002; Tsang and Vesilind, 1990; Lee and Hsu, 1995); dilatometry (Vesilind and Hsu, 1997; Smith and Vesilind, 1995); and centrifugal settling (Jin et al., 2004; Peng et al., 2011; Lee and Hsu, 1995). The method and conditions of these tests for the referenced papers are summarised in Table 2-2

Table 2-2. Methods for the determination of water distribution in sludge where P (kPa) is pressure, T (°C) is temperature, e (mm) is sample thickness, RH (%) is relative humidity, R_C (m) is centrifuge arm length, N (rpm) is rotational speed, t (s) is time and N.D is no data (adapted from Vesilind and Cezac 2004)

Method	Reference	Conditions
Drying Test		
Vacuum Drying	Smollen (1990)	P = 60kPa; T = 35°C; e = N.D.
Drying at Atmospheric Pressure	Tsang and Vesilind (1990) Lee and Hsu (1995) Chen *et al* (2002)	N.D. RH = 60.0%; T = 40°C; e = 1.0 mm RH = N.D.; T = 80°C; e = 0.0 mm
Dilatometry	Smith and Vesilind (1995) Vesilind and Hsu (1997)	Hydraulic oil; T = -35.0°C to -3.0°C Hydraulic oil; T = -8.0°C
Centrifugal Settling	Lee and Hsu (1995) Jin *et al* (2004) Peng *et al* (2011)	R_C= 0.135 m; N = 0-4000 rpm (0-2369g); t = 3600 s R_C= N.D.; N = 0-3000 rpm (0-1132g); t = 600 s R_C= N.D.; N = N.D. (0-2215g); t = 600 s

It has been observed in practice that results for tests on the water distribution in wastewater sludges depend largely on the method employed (Lee and Hsu, 1995; Jin et al., 2004). During drying tests, the drying of the sludge is greatly influenced by the operating conditions. Furthermore, analysis of the drying curves may be perturbed by shrinkage, sample thickness, crack and crust formation during drying. Therefore, the drying test "cannot measure the water distribution as an intrinsic parameter of the sludge" (Vaxelaire and Cezac, 2004). The main assumption for the dilatometric test is that the unfrozen proportion of water is representative of the bound water content. Smith and Vesilind (1995) found that the test was susceptible to air freed from the sludge during freezing, freezing temperature and solids concentration. In particular, Smith and Vesilind (1995) found that the precision of the method decreased with decreasing solids concentration. Vesilind and Hsu (1997) assumed that unfrozen water in sludge included bound water and some interstitial water but could not emphatically say at what point frozen water was only bound water. Thus the interpretation of results from dilatometric tests may be debatable (Vaxelaire and Cezac, 2004).

The basis of the centrifugal settling test is the assumption that, following centrifugation under an infinite rotational speed, a sludge sample tends towards an equilibrium height which corresponds to a sludge cake composed of only dry solids and bound water (Vaxelaire and Cezac, 2004). The bound water in this case is assumed to include intracellular water, surface water and interstitial water. Multiple difficulties have been found in the estimation of the equilibrium height as discussed in Vaxelaire and Cezac (2004) and the method has limited applicability to highly compressible media such as activated sludge. Jin *et al* (2004) suggested that the centrifugal settling test is more useful for determining sludge cake compressibility than bound water content. Equilibrium height has been found to vary linearly for rotational speeds between 500 and 3500 rpm (Lee and Hsu, 1995). However, the relationship becomes non-linear at increased speeds rendering the determination of equilibrium height by linear regression inaccurate (Jin et al., 2004). The determination of equilibrium height therefore requires rotational speeds equal to or less than 3500 rpm which may lead to a sizeable overestimation of bound water content.

Thus, as noted in Vaxelaire and Cezac (2004) representation of the water distribution in sludge is an interesting means of characterising the rate and extent of filterability. However, even if the water distribution could be accurately and consistently determined, "the solid-liquid separation is…controlled by numerous phenomena, and a single index is generally not sufficient to fully describe the whole operation" (Vaxelaire and Cezac, 2004). Other properties which affect the filterability of sludges are discussed in the following sections.

2.1.2 Solids Concentration

Vesilind and Hsu (1997) examined the effect of decreasing total solids content in sludge on the behaviour of bound water in aerobically digested sludges. As solids concentration decreased, bound water also decreased. Vesilind and Hsu (1997) concluded that at low solids concentrations, the sludge flocs were small and bound water consisted mainly of intracellular water and surface water. On the other hand, when the solids content was increased, larger flocs were formed trapping more water into which dissolved solids migrated and the bound water content increased. They considered the bound water to then include intracellular and surface water as well as some interstitial water.

Organic content and fractionation can significantly affect filterability of a sludge. Skinner *et al* (2015) compared the extent of dewatering of fifteen wastewater sludges including waste activated sludge, anaerobically digested sludge, trickling filter sludge, aged facultative lagoon sludge and sludge from a Cambi® reactor[4]. Assuming volatile suspended solids to be a surrogate for the sludge EPS content, Skinner *et al* (2015) found a strong relationship between volatile suspended solids and extent of dewatering for all sludges in the volatile solids range tested (from 40% to 80%). As the volatile suspended solids content reduced, final cake dry matter increased. Vincent *et al* (2012) compared results of drying septage and activated sludge on sludge drying reed beds (SDRBs) and found that septage had a greater extent of dewatering despite also having a higher organic content. They proposed that the discrepancy lay in the fractionation of the chemical oxygen demand (COD) – another measure of the organics content. The COD of the influent activated sludge consisted mainly of protein (44% COD_{total}) whilst for the septage influent it consisted mainly of fats (41% COD_{total}). As septage is stored for long periods of time, it contains a lower volume of easily degradable organics (protein = 17% COD_{total}) therefore the "deposits generated by activated sludge SDRB treatment retained more water than deposits produced by septage treatment" (Vincent et al. 2012). This result is in keeping with those reported regarding the role of EPS protein content in the water holding behaviour of sludge flocs. Ruiz *et al* (2010) and Ruiz and Wisniewski (2008) found that the more organic matter within urban residual sludge (a mixture of primary and secondary anaerobically digested sludge), the more difficult the sludge was to dewater.

As with wastewater sludges, the rate and extent of filtration of FS varies with initial total solids and volatile solids contents (Koné and Strauss, 2004). Pescod (1971) tested the drying

[4] The Cambi® reactor had a feed consisting of both waste activated sludge and anaerobically digested primary sludge (Skinner et al., 2015).

of septic tank sludge on USDBs with a sand and gravel filter. During dry season, Pescod (1971) observed that most water was lost through drainage in sludge with low initial solids content, whereas sludge with a high initial solids content drained slower and evaporation was more significant. Badji *et al* (2011) made a similar observation regarding thickened sludge deposited on an unplanted drying bed. Drainage was negligible and the main drying mechanism was evaporation. Semiyaga *et al* (2017) found that total solids as well as volatile solids and sand content[5] of FS from lined and unlined pit latrines from three different slums in Kampala City, Uganda, had a significant impact on the extent of dewatering. The extent of dewatering was taken as the cake dry matter content following centrifugation at 3000 rpm for twenty minutes. As sand content and total solids increased the sludge cake dry matter content increased, whilst increased volatile solids resulted in decreased cake dry matter content. Average cake dry matter content was much higher for FS from unlined pit latrines (31.8%) than for FS from lined pit latrines (18.6%) regardless of location. FS from unlined pits had a higher initial total solids content and sand content and volatile solids were lower. Similarly, Gold *et al* (2018) found that extent of dewatering was significantly affected by volatile solids and sand content for FS from septic tanks, unlined pits, lined pits and johkasou[6] tanks in Japan, Uganda and Vietnam. Gold *et al* (2018) also found strong correlation between extent of dewatering and COD.

In terms of the rate of filtration, Semiyaga *et al* (2017) observed little difference in CST values between FS from lined and unlined pits and did not report correlations with any sludge properties. This was despite significant differences in COD, total solids, total volatile solids and crude protein (which was used as an indicator of EPS content) between FS from lined and unlined pits. Average COD and total solids for FS from lined pit latrines were significantly lower than for FS from unlined pit latrines whilst average crude protein and volatile solids were higher for FS from lined pits. Gold *et al* (2018) found that FS from lined pits had significantly higher normalised CST values in contrast to most of the sludges tested apart from FS from lined pits in Japan and FS from septic tanks in Uganda. They did not find any significant differences in CST values for sludges from the same containment technology type even between countries. However, a similar trend was not observed when considering the extent of dewatering which was found to vary for sludges originating from the same

[5] Sources of grit and sand in pit latrine sludge is commonly a result of greywater entering the system which was used for cleaning utensils/vegetables, cleaning the home, showers located in the latrine superstructure, flooding and whether or not the pit is lined, partially lined or unlined (Strande et al., 2014)

[6] A johkasou tank is a domestic wastewater treatment system; the sludge may be more similar to wastewater sludge however, Gold *et al* (2018) class it as FS based on the fact that it is not connected to a sewer in accordance with the definition of FS by Strande *et al* (2014).

technology. Similar to results reported for wastewater, increased volatile solids and COD lead to increased CST and lower final cake dry matter.

2.1.3 Particle Size Distribution

Karr and Keinath (1978) suggested that particle size is the single most influential factor after testing the rate of dewatering of raw sludge, activated sludge and anaerobically digested sludge using specific resistance to filtration (SRF). SRF is a "measure of the resistance offered by a sludge to the withdrawal of water" (Karr and Keinath, 1978). A high SRF value implies a sludge which is difficult to dewater and vice versa. Fractionation was used to determine particle size distribution. Based on the sizing system shown in Table 2-3, Karr and Keinath (1978) concluded that supra-colloidal particles (1.0 μm to 100.0 μm) had the greatest influence on the rate of sludge dewatering. For all the sludges tested, the higher the proportion of supra-colloidal particles, the higher the SRF value. Karr and Keinath (1978) also concluded that changes in dewaterability attributed to pH, biological degradation, mixing and conditioning could be explained by considering the affect each parameter had on the particle size distribution of the sludge.

Table 2-3. Solids classification system (Karr and Keinath, 1978)

Solids Fraction	**Size (μm)**
Settleable	≥100
Supra-colloidal	1-100
True colloidal	0.001-1
Dissolved	≤0.001

Similarly, Smollen (1986) found that the rate of dewatering of sludge was affected by the fine solids concentration. The SRF and capillary suction time (CST) were determined for six types of municipal wastewater sludge: primary sludge; return activated sludge; waste activated sludge thickened by dissolved air flotation; anaerobically digested primary sludge; anaerobically digested mixture of primary; and activated sludge and sludges heat-treated by the Zimpro process which uses wet air oxidation. CST is another, more commonly used measure of rate of dewatering. Sludges with a low CST will dewater easily and vice versa. Smollen (1986) found that greater percentage fines led to higher CST and SRF values i.e. slower rate of dewatering. Similarly, Shao *et al* (2009) found that decreasing particle size resulted in higher CST values; all of the particles in the sludges tested fell into the supra-colloidal category shown in Table 2-3. The SRF, CST and other measures of dewaterability are discussed in more detail in Appendix B.

Particle size distribution has also been found to influence the dewaterability of FS. Vincent *et al* (2011) compared the performance SDRBs in the treatment of activated sludge and septage. High CST values were found for septage due to the high proportion of fine particles present which resulted in a larger surface area and greater water retention. CST values of septage increased with decreasing d_{10} (grain sizes that are 10% finer by volume). However, Vincent *et al* (2011) concluded that the composition of the septage appeared to have a greater impact on filterability, as samples with the same d_{10} gave markedly different CST values. Semiyaga *et al* (2017) found that the main proportion of solids in FS from both lined and unlined pits was below 32µm, within the supra-colloidal range as defined by Karr and Keinath (1978). A greater percentage of particles in FS from unlined pit latrines was less than 32µm implying a greater proportion of fines present. There were also greater proportions of all other particle sizes in FS from unlined pits (5.0 mm to 1.0 mm; 1.0 mm to 75.0 µm; 75.0 µm to 32.0 µm). The greater percentage fines may have been the cause of the slightly higher CST values observed, whilst the greater total solids proportion would have contributed to a higher percentage cake dry matter.

2.1.4 Flocculation and Extrapolymeric Substances

The formation of flocs or flocculation is the process by which microorganisms and suspended or colloidal solids in a sludge form larger aggregates (Wilen, 1995). It is critical to the optimum operation of the treatment process and determines how a sludge will dewater, flow and settle (Nguyen et al., 2007; Sanin and Vesilind, 1999; Karr and Keinath, 1978). Flocculation is the main component of the activated sludge process. Activated sludge "is an aerobic, suspended growth process, in which microorganisms are grown in a variety of bioreactor configurations for the purpose of removing soluble organic matter" (Nguyen et al., 2007). The process is routinely used for the treatment of municipal and industrial wastewater. An activated sludge floc – shown in Figure 2-2– is defined by Sanin and Vesilind (1999) as "a microbial entity that is formed by different species of bacteria as well as other organisms like protozoa, fungi, algae and viruses along with some suspended abiotic material" which are embedded in a polymeric network consisting of humic-like substances, polysaccharides, proteins, lipids and nucleic acids known as extracellular polymeric substances (EPS). The polymeric network thought to be the result of secretions by microorganisms present in the sludge (Das et al., 1993).

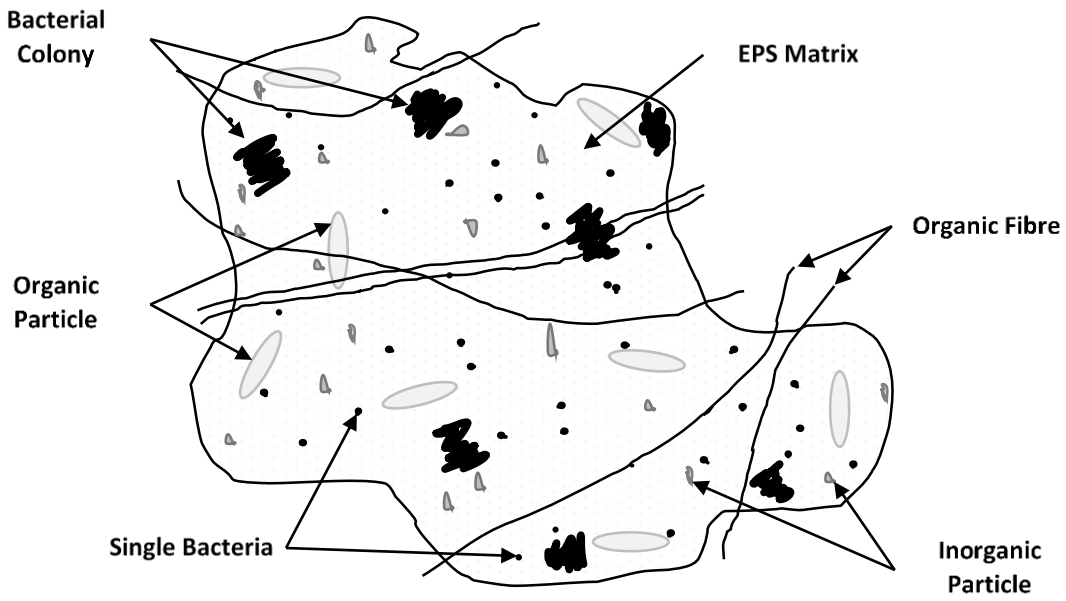


Figure 2-2. Schematic structure of activated sludge floc (Alrhmoun, 2015)

Sludge flocs – their size, shape, density and the strength of floc structure – have a profound effect on the settling and dewatering properties of wastewater sludge (Sobeck and Higgins, 2002). EPS has a significant impact on sludge floc properties since it makes up 40% to 60% of the total dry matter of a floc (Christensen et al., 2015). Therefore, it may be inferred that EPS content would have a significant effect on rate and extent of sludge dewatering. Mikkelsen and Keiding (2002) found that increased total EPS content had a stabilising effect on floc structure in terms of shear sensitivity, degree of dispersion[7] and floc size. This led to faster rates of dewatering as measured by SRF than sludges with a low EPS content. Mikkelsen *et al* (2002) considered EPS to have a "water-holding capacity, a term introduced to cover both osmotic water as well as water physically trapped within the polymer network". A large proportion of the interstitial water in the polymer network may be held by osmotic pressure within the sludge flocs due to the presence of EPS which is negatively charged. Mikkelsen and Keiding (2002) found that the EPS humic[8] content and EPS protein content were the main contributors to EPS charges. The EPS protein content was also related to the bound water content as it "could have a particular stabilising effect on the sludge structure, affecting the water entrapment in the sludge" (Mikkelsen et al., 2002). A similar result was observed by Jin *et al* (2012) with regards to the contribution of EPS protein content to the water binding ability of sludge flocs. Jin *et al* (2012) also found a significant contribution to water binding ability from EPS carbohydrate content but noted little effect from EPS humic content.

[7] Degree of dispersion is a measure of the colloidal dispersion in the sludge.

[8] Humic substances in wastewater are similar to those present in sols (Riffaldi et al., 1982).

EPS may be further defined as possessing a multi-fractioned structure consisting of soluble EPS (or slime), loosely bound EPS and tightly bound EPS (Shao et al., 2009; Christensen et al., 2015). Following the removal of EPS from the sludge floc, Shao *et al* (2009) referred to the cells in the residues as a pellet. Shao *et al* (2009) found that protein was the primary component of the sludge flocs, concentrated in the pellet with a small proportion in the tightly bound EPS. Shao *et al* (2009) also found that an increase in soluble organic matter (predominantly protein in the soluble ESP) resulted in higher CST values but the increase of protein in the pellet could result in lower CST values i.e. faster rate of dewatering. This is logical when the presence of protein in the soluble ESP is a result of floc degradation. Shao *et al* (2009) observed that the proportion of protein in the pellet decreased but increased in the tightly bound EPS and soluble EPS fractions under alkaline conditions which cause the breakdown of sludge flocs. Thus as flocs broke down, the protein content appeared to transfer from the inner fraction of the sludge floc (pellet) to the outer fractions (Shao et al., 2009). It would appear that high EPS content results in more bound water and therefore a low EPS content is desirable with regards to the achievement of high cake solids content following filtration (Mikkelsen and Keiding, 2002; Mikkelsen et al., 2002). However, if the aim is for the sludge to dewater quickly then a high EPS content would appear to be beneficial.

EPS content has been found to play a different role in the dewatering of FS. Ward *et al* (2019) measured the EPS content of FS from a variety of sources including schools, public toilets, offices, restaurants and houses of worship in Dakar, Senegal and Dar es Salaam, Tanzania. Containment technology included septic tanks, pit latrines, one cesspit and one septic tank effluent storage pond. EPS content in FS was found to be much lower than values reported for activated sludge and were only slightly more comparable with primary sludge or mesophilic anaerobically digested sludge. This is unsurprising as FS is more likely to undergo anaerobic digestion during storage which breaks down the polymer network as discussed previously. FS originating from public toilets contained a higher proportion of EPS indicating it was less stabilised, potentially due to shorter storage times or due to higher levels of ammonia[9] which would inhibit anaerobic bacteria and decomposition even for longer storage times (Ward et al., 2019). Even in less stabilised, type A sludges the EPS content was considered to be too low to promote flocculation as it does in wastewater sludge. Instead, Ward *et al* (2019) found that EPS content, especially EPS humic content, was strongly positively correlated with CST and turbidity. This suggested that EPS in FS could "be more accurately described as colloidal and suspended organic matter that contributes to filter

[9] Usage behaviour in public toilets may lead to a significant increase in urine which has been shown to lead to extremely high ammonia concentrations (Ward et al., 2019).

blinding and turbidity, as opposed to polymeric glue that binds flocs together" (Ward et al., 2019). Ward *et al* (2019) also concluded that EPS content and volatile suspended solids were not as significant an indicator of the extent of dewatering as they are for wastewater sludge.

2.1.5 Ionic Composition and Concentration

Ionic composition and concentration as described by pH, conductivity and water hardness has been found to have a significant impact on floc structure and strength due to the alteration of cations present in the sludge (Christensen et al., 2015). This is due to the negative charge of the sludge flocs (because of the negatively charged EPS). As outlined by Sobeck and Higgins (2002), the role of cations in flocculation may be described by several theories: Double Layer (DVLO) Theory; Alginate Theory; Divalent Cation Bridging Theory (DCB). Sobeck and Higgins (2002) concluded that the DCB Theory best described the long-term effect of cation changes on floc properties and noted that DLVO Theory seemed only applicable to short term effects. A similar result was found by Mikkelsen and Keiding (2002). High concentrations of multivalent cations like calcium and magnesium results in strong, compact flocs and good rate of dewatering. High proportions of monovalent cations – which cause high conductivity – such as sodium and potassium have a degradative effect on floc strength resulting in the breakdown of the flocs (Christensen et al., 2015; Sobeck and Higgins, 2002). Floc breakdown leads to the release of fine particles which can cause slower dewatering as discussed previously. It would also likely result in increased cake dry matter content indicating greater extent of dewatering. Sobeck and Higgins (2002) found that the addition of calcium increased the cake dry matter content more than the addition of magnesium. This finding was potentially because calcium ions reduced the bound water content, but magnesium ions did not (Sobeck and Higgins, 2002).

The pH of a sludge has been found to affect the charge of EPS and thus affects floc structure and the dewaterability of sludge indirectly (Christensen et al., 2015). Shao *et al* (2009) found that under alkaline conditions sludge dewatering rate was greatly reduced due to the release of organic matter (mainly protein and polysaccharides) from sludge flocs. Although Shao *et al* (2009) also observed that dewatering rate only slightly improved under acidic conditions. Similarly, Raynaud *et al* (2012) found that SRF increased significantly under alkaline conditions but there was little difference between pH = 3 and pH = 7. Raynaud *et al* (2012) also found that, although acidic conditions did not appreciably modify the dewatering rate, they resulted in a higher cake dry matter. The main effect of pH appears to be with regards to the charge of EPS content of a sludge floc. At neutral (pH = 7), EPS components have a negative charge whilst at low pH they have almost no charge and at high pH the charge increases (Christensen et al., 2015). At high pH values, the polymer network breaks down due

to changes in charge thus releasing fine particles into suspension affecting rate and extent of dewatering. Karr and Keinath (1978) found that a decrease in pH improved SRF of activated sludge as pH altered the particle size distribution; the proportion of supra-colloidal particles decreased with decreasing pH.

In FS, turbidity and CST have been found to positively correlate to high pH, conductivity and a high proportion of monovalent cations (Ward et al., 2019). Gold *et al* (2018) also found that increased conductivity and ammonia nitrogen i.e. an increase in monovalent cations, had the strongest correlation to CST and led to decreased rate of dewatering. Ward *et al* (2017) found that urine concentration – which impacts ammonia concentration – had "no discernible impact on the absolute dewatering rate but did impact normalised CST and dewaterability." This may have been due to the impact urine concentration has on total solids content: high urine concentrations were found to increase total solids both before and after dewatering. However, in light of the findings by Gold *et al* (2018) it is likely that a higher ammonia concentration due to higher urine concentration would be to blame. It was noted that high urine concentrations "decreased the efficiency of centrifuge dewatering for both low- and high-strength synthetic sludge" (Ward et al., 2017).

2.1.6 Type of Sludge and Degree of Digestion/Stabilisation

The degree of digestion is a key factor in determining the ease with which sludge may drain as it impacts a number of sludge properties already discussed. Smollen (1986) found that the SRF increased significantly following anaerobic digestion of both primary sludge and activated sludge. Smollen (1986) concluded that the difference in floc structure between anaerobic sludge flocs and activated sludge flocs was the reason for the increased SRF following anaerobic digestion. Mikkelsen and Keiding (2002) also found that activated sludge had better dewaterability than primary and anaerobically digested sludges. However, the primary and anaerobically digested sludges produced a higher cake dry matter following filtration. The disparity was likely caused by the low EPS content of the primary and anaerobically digested sludges. This is similar to the conclusion drawn by Smollen (1986) with regards to difference in floc structure. Houghton *et al* (2001) investigated the effects of increasing EPS on CST values of activated sludge, raw sludge and digested sludge. Activated sludge was found to have the highest proportions of EPS and lowest CST values whilst raw sludge and digested sludge had higher values of CST and lower proportions of EPS. These results are in keeping with those of Smollen (1986) and Mikkelsen and Keiding (2002). However, Houghton *et al* (2001) also found that there appeared to be an optimum EPS content for all three types of sludge and this optimum proportion decreased for each sludge type. After this optimum EPS content, they found CST values increased with increasing EPS

content. Although, Houghton *et al* (2001) did not report EPS fractions, nor did they report the different flocculation behaviour of the different types of sludge.

Kopp and Dichtl (2001) noted that digestion also affects particle size distribution in a sludge. During digestion, particle size decreases leading to an increase of interstitial water. Kopp and Dichtl (2001) found that the effect of particle size appeared to be dominant but due to the spread of measured values, the dewaterability of a sludge cannot be determined from particle size distribution alone (Kopp and Dichtl, 2000). Houghton and Stephenson (2002) looked at the effect of varying organic composition on the EPS content and composition, particle size and CST of anaerobically digested primary sludge. Digestion of the primary sludge and primary sludge with the addition of glucose of propionic acid was found to increase CST for all sludges but suspended solids were reduced. CST of the primary sludge and glucose was almost double that of primary sludge and primary sludge with propionic acid. An increase in the organic composition of EPS was found in digested primary sludge with glucose whilst it decreased in digested primary sludge with propionic acid. Digestion of all sludges resulted in an increase in EPS protein content and decrease in EPS carbohydrate content. Increase in CST was related to higher proportions of EPS and an increase in mean particle size whilst the composition of EPS was not found to have an effect. As was observed by Houghton *et al* (2001) the increase in CST with increase in EPS was noted past a certain threshold value. No conclusions were drawn as to the impact of EPS content on the extent of dewaterability. It would appear that an increase in EPS results in a decrease in CST up to a threshold value after which point a further increase will result in an increase in CST.

Degree of digestion has also been found to impact the proportion of volatile solids (organics) present in sludge. During digestion the proportion of volatile solids decreases which results in less bound water and greater cake dry matter content after filtration. Houghton and Stephenson (2002) found that the volatile solids of the three sludges tested reduced following anaerobic digestion. The greatest reduction was recorded for the mixture of primary sludge and glucose. Theoretically, the greater the proportion of organic compounds present in a sludge, the lower the density of the sludge particles ($\rho_{organic} \approx 1.0$ g cm^{-3}; $\rho_{inorganic} \approx 2.3$ g cm^{-3}) and the greater the volume of bound water. This is because the contact angle of organic particles is smaller than that of inorganic particles, thus binding more water with capillary forces (Kopp and Dichtl, 2001b). Results from Kopp and Dichtl's (2000) research showed a tendency for the achievable cake dry matter content to increase with decreasing volatile solids content. Conversely it would be expected that an increase in volatile solids would result in an increase in rate of filtration.

Type B sludge has been found to dewater more easily than type A sludges which are less stabilised i.e. digested. During a pilot study on USDBs in Accra, Ghana, Heinss *et al* (1998) found that type A sludge applied to a USDB exhibited erratic results: some batches did not dewater at all whilst others dewatered to a total solids content of 29%. The variation was likely due to differences in age – and thus stabilisation – of the sludges tested (Heinss et al., 1998). Cofie *et al* (2006) attributed the variation in drying time of sludge to achieve a total solids content of 20% on a USDB with sand and gravel filter to the varying degree of stabilisation of different sludges. Ward *et al* (2017) found that high strength synthetic FS i.e. type A, had reduced dewatering rates when compared with low strength synthetic FS i.e. type B. Ward *et al* (2019) found that public toilet sludge had higher CST values and salt concentrations than other sludges but noted that there was little difference in cake dry matter between different sources. This would imply that the type of sludge and thus the degree of digestion could have a significant impact on the rate of dewatering of FS but perhaps not on the extent of it. The type of FS is significantly affected by type of onsite technology used, storage duration, inflow and infiltration, local climate, collection method and whether or not additives or greywater are added to the system (Schoebitz et al., 2014; Strande et al., 2018).

2.2 Sludge Drying Behaviour

Generally, two or three phases of drying may be observed in the drying of wastewater sludges and FS. These phases tend to be described according to the consistency of the sludge or according to the rate of evaporation. Gruter *et al* (1990) observed three distinct zones of consistency of sludge dried in a thin film sludge dryer: the wet sludge zone in which sludge is similar in consistency to a free-flowing pulp; the stickiness zone in which the sludge behaves similarly to a thick paste; and the granules zone in which the sludge becomes crumbly. Similarly, Ferrasse *et al* (2002) identified a pasty stage, a lumpy stage and a granular stage for indirect agitated sludge drying. Alternatively, wastewater sludge has been observed to exhibit a brief induction (warm-up) stage, a constant rate stage and one or two falling rate periods which may be observed using drying curves. Similar behaviour has been observed in FS drying (Pescod, 1971; Makununika, 2016)

A drying curve is often used to show the evolution of average moisture content within a particular material dried under specific conditions (Vaxelaire and Puiggali, 2002). Importantly, they describe "the product response, not the external conditions as provided by the drying equipment" (Hall, 1987). As shown in Figure 2-3, three types of drying curves may be used to present data from drying tests of wastewater sludge:

- a drying curve of moisture content versus time obtained directly from mass loss-time data;
- a drying rate curve of drying rate versus time which is the differential of the drying curve.;
- a Kirscher curve of drying rate versus moisture content, the Kirscher plot is most often derived by combining the drying and drying rate curves (Kemp et al., 2001).

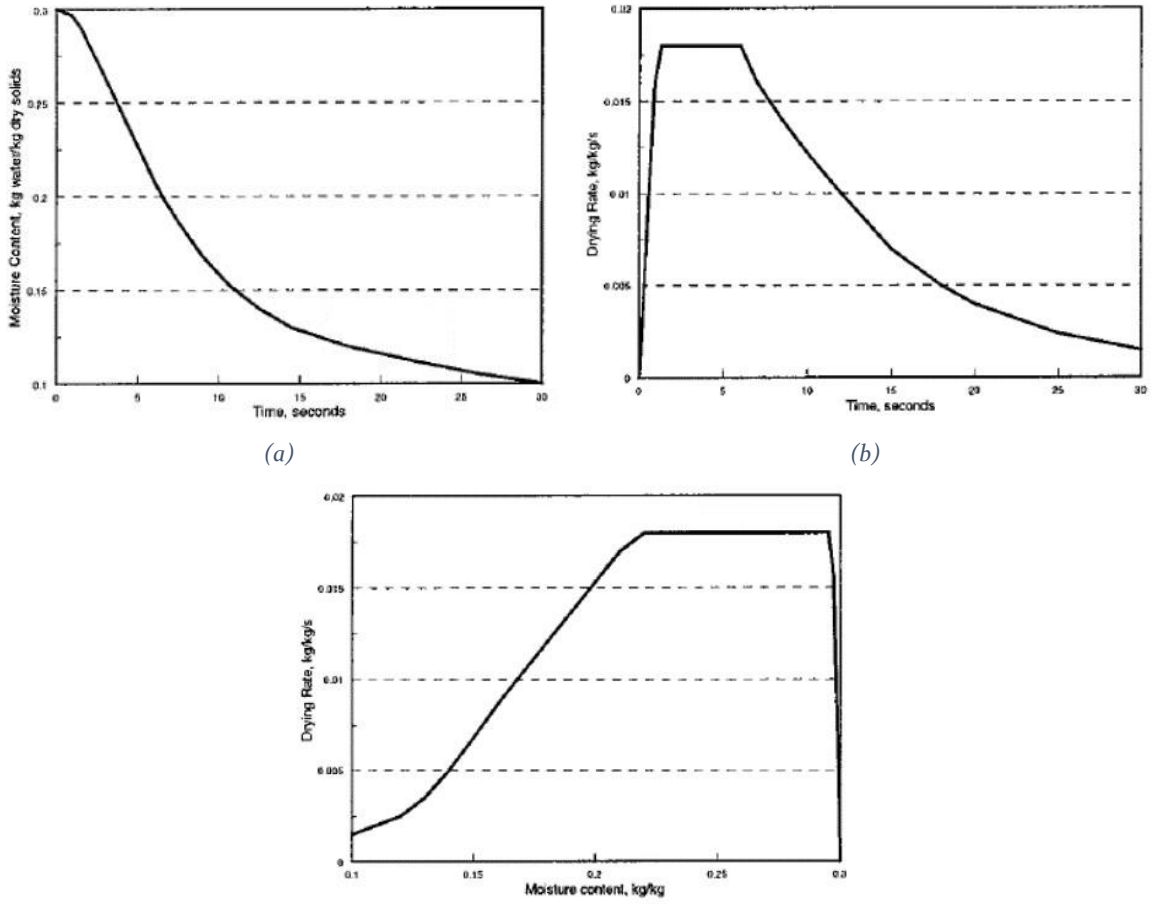


Figure 2-3. Drying curves for a particle dried in a crossflow dryer showing an induction (initial heating) stage, constant rate stage and falling rate stage (Kemp et al., 2001)*: (a) moisture content-time plot; (b) drying rate-time plot; (c) drying rate-moisture content (Kirscher) plot*

During the constant rate stage of drying, evaporation is primarily controlled by external factors: radiation, air temperature, relative humidity and wind speed. The Kirscher plot in Figure 2-4 shows the evolution of drying of an activated sludge sample. There is a brief induction period followed by a constant rate period in which it is assumed that only free water evaporates from the surface of the material. As the drying boundary progresses into the sludge, the falling rate period of drying begins at the first point of inflection of the graph marked as W_{cl} or critical moisture content. This point on the graph is often used to estimate the amount of bound water present in the drying material (Vaxelaire and Cezac, 2004). Since the moisture content at W_{cl} is dependent on both the substance being dried and the external conditions it should not be considered an intrinsic parameter of the sludge (Vaxelaire and

Cezac, 2004). As illustrated in the boxes in Figure 2-4, below the drying boundary – marked by the horizontal black dotted line – free water migration exists, whilst above it only bound water (in the case of a hygroscopic material) and vapour are removed (Vaxelaire and Cezac, 2004). When drying progresses, the heat and mass transfer resistances increase, and the rate of evaporation consequently decreases. A second falling rate period may also be observed beginning at the second point of inflection on the graph, marked as W_{c2}, after which the graph tends towards the equilibrium moisture content marked as W_{eq}. This point on the graph may be used to identify the amount of intracellular water present in a sludge (Vaxelaire and Cezac, 2004; Tsang and Vesilind, 1990; Kopp and Dichtl, 2001b; Kopp and Dichtl, 2000; Smollen, 1990; Lee and Hsu, 1995). However, as mentioned previously, drying curves are useful only to provide an approximation of moisture distribution in a sludge under a given set of external parameters.

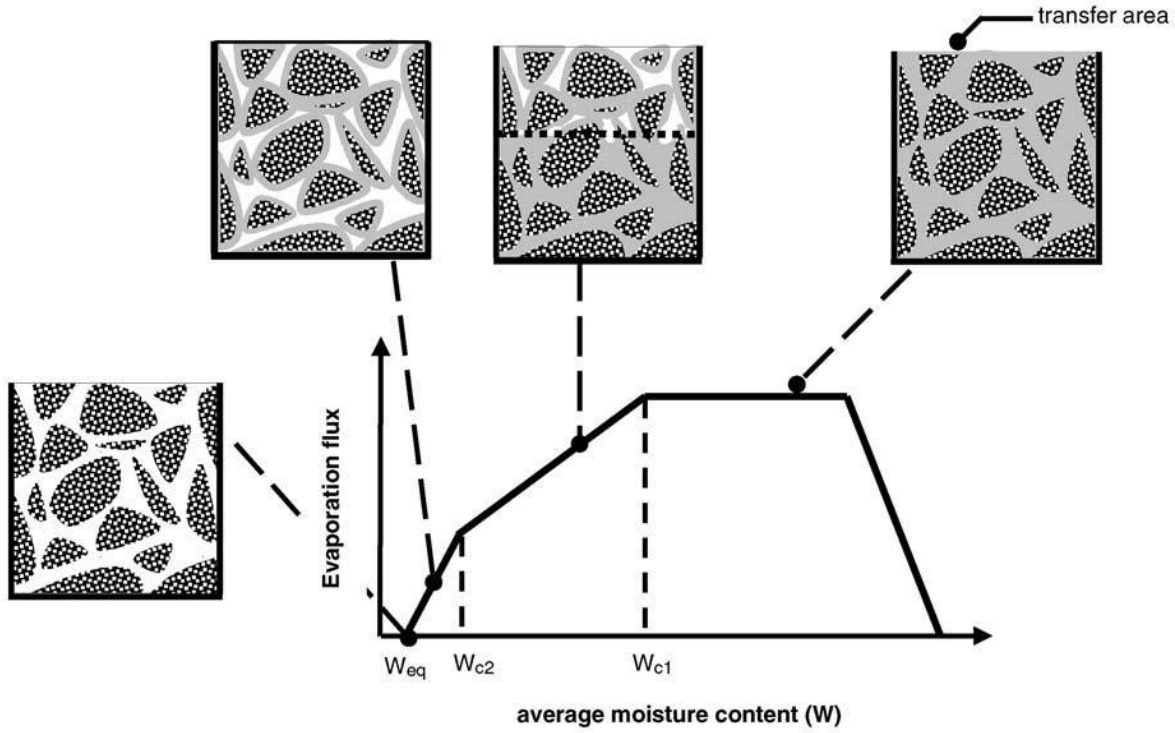


Figure 2-4.Classical Kirscher plot for activated sludge where W_{c1} is the water content at the first transition point on the drying curve also called the critical moisture content W_c, W_{c2} is the water content at the second transition point on the drying curve and W_{eq} is the equilibrium moisture content (Vaxelaire and Cezac, 2004)

During the drying of both wastewater sludge and FS it is common to observe crust formation, shrinkage, and cracking. The impact of these phenomena on the drying rate of sludge is discussed in the following sections.

2.2.1 Crust Formation

Crust formation can retard the drying rate in sludges as it reduces the ability of the surface of the sludge to transmit moisture to the air (Tao et al., 2005). It may also result in the absence

of a constant rate drying period (Vaxelaire and Puiggali, 2002). When testing sludge samples in a solar drying rig, Septien, Mugari *et al* (2018) found that increased air velocity resulted in decreased drying rate, a similar result was observed with increase in moisture content from 30°C to 60°C. These results were thought to be due to crust formation which was also observed in drying experiments with wastewater sludge. Septien, Mugauri *et al* (2018) hypothesized that increasing the air velocity or temperature led to faster drying of the sludge surface layer resulting in faster crust formation which would hinder further drying. They also noted that crust formation did not occur in overcast conditions and so hypothesised that the "combination of high temperature with solar irradiance could…have a structural effect on crust formation, making it harder" (Septien, Mugauri, et al., 2018).

Crust formation may be reduced with the addition of rice straw or sawdust as discussed in the following section, or with the addition of artificial cracks. Tao *et al* (2006) found that crust formation on the surface of dewatered anaerobically digested primary and waste activated sludge cakes was altered due to the addition of artificial cracks. In cakes without artificial cracks the crust was observed to form at the border of the sludge cake and progress towards the centre. For cakes with initial cracks in the surface the crust was observed to form between cracks and move out towards the border of the sludge cake.

2.2.2 Shrinkage

As shrinkage affects the transfer area of the sludge it affects the drying rate. According to Leonard *et al* (2002) shrinkage is "a key phenomenon which controls both the drying rate, the final texture and mechanical properties of the dried material." According to Vaxelaire and Puiggali (2002) shrinkage of sludge pellets (both height and diameter) was found to be significant over the course of the drying experiments in both PVC and activated sludges. Vaxelaire and Puiggali (2002) observed that the shrinkage ratio decreased linearly to a low moisture content after which point it maintained a constant value. Similarly, Leonard *et al* (2002) found that shrinkage of secondary wastewater sludges decreased linearly with residual water content down to a critical threshold. Leonard *et al* (2002) also found a maximum sludge volume reduction of 80% showing the key role shrinkage plays in controlling the drying process.

In order to weaken the influence of shrinkage and crust formation during convective drying of dewatered wastewater sludge, Cai *et al* (2015) investigated the effect of adding rice straw to sludge cakes. During drying of pure sludge Cai *et al* (2015) observed the formation of a thick crust on the surface whilst the centre remained wet, as observed by Vaxelaire *et al* (2000) and Vaxelaire and Puiggali (2002). However, they found that the addition of 1%, 2%

and 3% rice straw retarded shrinkage and reduced the crust formation on the surface of the sludge but did not limit cracking thus improving sample drying rates. Minimal shrinkage was noted in the sludge with 5% rice straw but Cai *et al* (2015) concluded that due to the low thermal conductivity of the rice straw (less than 0.1 W m^{-1} K^{-1} at 160°C) the thermal conductivity of the sludge overall was decreased so the drying rate was lower. It was also noted that fewer cracks appeared in the sludge samples with a higher percentage of rice straw, likely due to the rice straw acting as a filler and providing a 'skeleton' for the sludge, inhibiting both shrinkage and cracking. The latter would also result in a reduced drying rate.

Li *et al* (2016) investigated the effect of adding sawdust to dewatered wastewater sludge for convective drying. It was expected that the addition of sawdust would act in the same manner as rice straw by "reinforcing the texture of soft and pasty sludge" (Li et al., 2016). This would result in an increased drying rate by reducing shrinkage. Following drying experiments Li *et al* (2016) found that the drying time did decrease with increasing ratio of sawdust, resulting in a higher average drying rate when compared with pure sludge. Increasing the ratio of sawdust to sludge had "the effect of bracing the structure of the sludge… resulting in smaller shrinkage and higher total exchange surface of the sludge bed during the drying process, which is beneficial for sludge drying" (Li et al., 2016). It was also observed that both the initial water content and the total amount of water removed decreased with the addition of the sawdust likely due to absorption of water by the sawdust, but no final water contents or total solids contents were reported.

2.2.3 Cracking

It is desirable to reduce shrinkage and crust formation during drying of sludge, but cracking is considered to result in increased drying rates. Tao *et al* (2005b) observed that the peak drying rates of dewatered anaerobically digested primary and waste activated sludge cakes with artificial cracks were higher than for cakes with no cracks. Leonard *et al* (2002) found that cracks in the surface of mechanically dewatered wastewater sludge appeared at the end of the drying process. Thus the authors concluded that the onset of crack formation was related to the appearance of internal transfer mechanisms (i.e. humidity and temperature gradients within the sludge). However, during convective drying of dewatered anaerobically digested primary and waste activated sludge cakes, Tao *et al* (2005a) observed that shrinkage and crack formation occurred simultaneously during drying. The overall drying rate for the artificially cracked sludge cake was found to be higher. Marklund (1997) found that the drying rate of thick or thin layers of aerobically digested activated and chemical wastewater sludge was independent of sludge surface cracking until the total solids content reached approximately 30% or more.

Cracking is often observed during FS treatment on both covered and uncovered USDBs with and without sand filters as shown in Figure 2-5. However, Septien, Mugari *et al* (2018) observed little to no cracking when testing FS samples in a solar drying rig during overcast conditions. Despite associating cracking with external conditions, limited cracking was also observed on FS samples during open-air drying in both sunny and cloudy conditions. No literature was found detailing the potential impact of internal sludge properties on cracking.

(a)

(b)

Figure 2-5. Cracking of faecal sludge dried on unplanted sludge drying beds: (a) freshly loaded and partially dewatered faecal sludge on unplanted sludge drying beds at Niayes faecal sludge treatment plant, Dakar, Senegal (Strande et al., 2014)*; (b) covered, unplanted sludge drying bed with sand filter at Buro Happold Engineering pilot site in Faridpur, Bangladesh* (Way, 2013a)

2.3 Key Properties Affecting the Drying of Sludges

Little research was found detailing the effect of the characteristics of sludge on its drying. Two reasons as to why this may be the case are as follows:

1. When sludge drying is considered, it is often in terms of how the operational parameters known to affect the rate of evaporation i.e. air velocity, air temperature and relative humidity affect the drying rate of sludge.
2. Often the purpose for sludge drying is to determine the effect of drying on the properties of the sludge rather than the effect of the properties on drying rates (see Afolabi *et al* (2014); Makununika (2016); Septien, Singh *et al* (2018); Septien, Mugari *et al* (2018).

Therefore, in this section the key properties which have been found to impact the rate of evaporation from sludge or which are likely to affect the rate of evaporation from sludge are briefly outlined. Following this a section regarding the effect of operational parameters on the drying of FS is included due to the relevance for the development of drying tests in Chapters 5 and 6 of this thesis.

2.3.1 Moisture Distribution

Moisture distribution is as relevant for evaporation as for filtration as it is assumed that bound water offers greater resistance to evaporation (Lee and Hsu, 1995). Therefore, a high proportion of bound water content may prolong the total drying time and affect the final total solids content, much like in filtration. It will also affect the amount of energy necessary to achieve a desired final total solids content.

2.3.2 Particle Size Distribution

Vaxelaire and Puiggali (2002) investigated the effect of particle size on the drying kinetics (i.e. drying behaviour) of PVC sludge. As shown in Figure 2-6, for the same drying potential, decreasing the particle size of the PVC sludge resulted in the modification of the position of the transition between the constant rate and falling rate phases (Vaxelaire and Puiggali, 2002). It was also concluded that the drying kinetics were slower for particles with a larger diameter which was taken to imply that "both the rate of drying and the shrinkage are dependent on the particles size" (Vaxelaire and Puiggali, 2002).

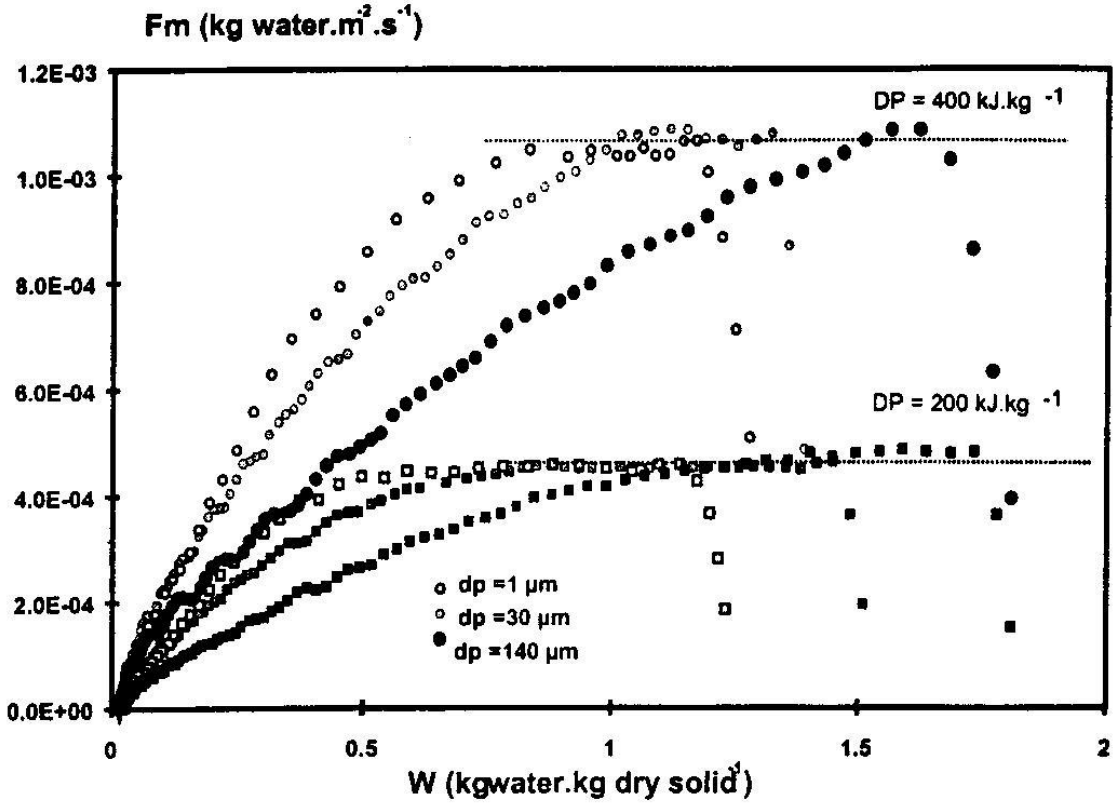


Figure 2-6. The particle size effect on the drying kinetic where DP is the drying potential, W (kg water kg dry solid^{-1}) is the moisture content and Fm (kg water m^{-2} s^{-1}) is the evaporation mass flux density and dp is the particle size (Vaxelaire and Puiggali, 2002)

2.3.3 Type of Sludge

Vaxelaire and Puiggali (2002) found that a constant rate period was not observed for activated sludge during drying tests, even at high moisture contents. This was assumed due to the formation of a crust during drying. For PVC sludge no crust was observed. Due to its porous structure, the mass transport of water from within the PVC sludge was not overly affected and the external drying conditions largely controlled the drying process (Vaxelaire and Puiggali, 2002). These findings highlight the significance of the affect that the type of sludge can have on its drying behaviour. Figure 2-7 illustrates this point further and shows typical drying curves for primary sludge, raw sludge and secondary sludge.

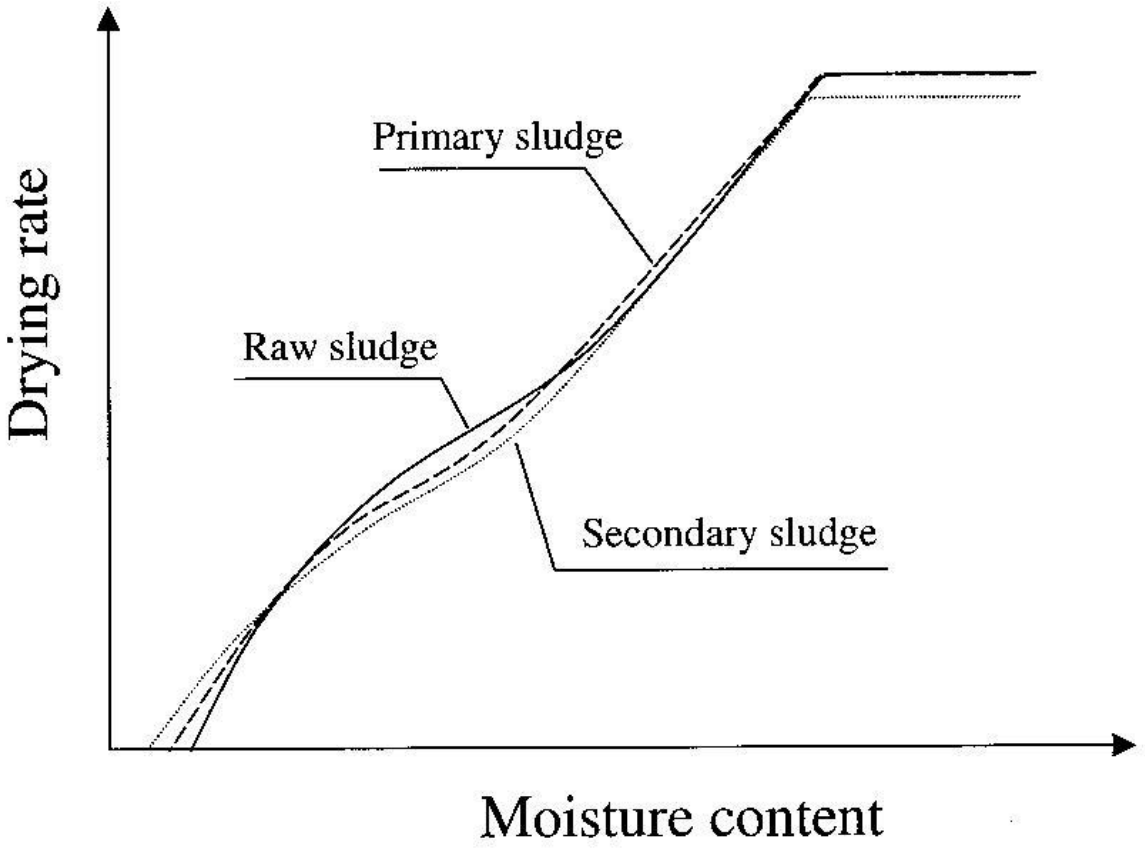


Figure 2-7. Typical drying rate curve for different wastewater sludges (Chen et al., 2002)

2.3.4 Thermal Properties

Some thermal properties (thermal conductivity, and specific heat capacity) of sludge have been reported in several instances; however, this was to assess the impact of drying on them rather than the impact that they may have on drying rates. However, it is likely that thermal properties will have some impact on the rate of evaporation from sludge and therefore merit some discussion. This is because the sensible heat flux, G (J m^{-2} s^{-1}), i.e. the conduction into and out of an evaporating body, cannot be ignored.

As no discussion regarding thermal properties was found in the literature encompassing wastewater sludge and FS, this discussion is taken from the field of soil physics. It is assumed that there will be some similarity in the behaviour of soils and sludges given that both can be highly organic, triphasic materials. Thermal conductivity may be defined as "the amount of heat passing in unit time through a unit cross-sectional area of [a body] under a unit temperature gradient applied in the direction of this heat flow" (Farouki, 1986). Specific heat capacity may be defined as the quantity of heat required to raise a unit weight of material by 1°C (Kersten, 1949). Other thermal properties include the volumetric heat capacity and thermal diffusivity. Volumetric heat capacity is related and may be defined as "the change in heat content of a unit bulk volume of soil per unit change in temperature" (Hillel, 1998).

Thermal diffusivity is the ratio of thermal conductivity and specific heat capacity and governs the thermal behaviour of a soil in the existence of an unsteady state (Farouki, 1986; Boguslaw and Lukasz, 2003). A high value for thermal diffusivity implies a capability for rapid, large temperature changes (Farouki, 1986). Thermal diffusivity is assumed to be affected by those properties which affect either the specific heat capacity and thermal conductivity and therefore it is not discussed exclusively. According to Hanson *et al* (2000) the four parameters are related as follows:

$$\alpha = \frac{\lambda}{C} = \frac{\lambda}{c\rho}, \tag{2.2}$$

where α ($m^2\ s^{-1}$) is the thermal diffusivity, λ ($W\ m^{-1}\ °K^{-1}$) is the thermal conductivity, C ($J\ m^{-3}\ °C^{-1}$) is the volumetric heat capacity, c ($J\ kg^{-1}\ °C^{-1}$) is the specific heat capacity and ρ ($kg\ m^{-3}$) is the density.

Thermal properties are highly dynamic, meaning that they change with time and based on both material properties and external factors. Temperature has been found to have a significant effect on the thermal properties of soils. Kersten (1949) found that both specific heat capacity and thermal conductivity for various soils increased with increasing temperature. Nusier and Abu-Hamdeh (2015) observed a similar phenomenon for the specific heat capacity and volumetric heat capacity of sand and a loam soil.

The moisture content of a soil also has a significant effect on its thermal properties (Becker et al., 2013; Dong et al., 2015; Salomone et al., 1984; Hanson et al., 2000). At constant dry bulk density, thermal conductivity has been observed to increase with an increase in moisture content (Kersten, 1949). Nusier and Abu-Hamdeh (2015) observed that both specific heat capacity and volumetric heat capacity increased with increasing moisture for a sand and a loam soil at constant density due to the high specific heat capacity of water. Assuming an increase in moisture content results in a decrease in volume of voids filled with air, Rubio (2013) found that both thermal conductivity and specific heat capacity decreased as the volume fraction of air in silt loam soils increased. As moisture is added to the soil a thin water film develops bridging, and then filling, the voids between soil particles expelling air and increasing the heat flow, resulting in higher thermal conductivity (Becker et al., 2013; Salomone and Kovacs, 1984; Salomone et al., 1984). This phenomenon occurs slowly at first and then more rapidly as the moisture fills the gaps between particles; when all the voids are filled an increase in moisture content will no longer increase heat flow and the thermal conductivity of the soil doesn't significantly increase. As the soil dries and the moisture content decreases, the process occurs in reverse. Moisture is lost from soil pores and replaced with air so the thermal conductivity and volumetric heat capacity decrease (van Bavel and

Hillel, 1976). Eventually the 'thermal bridge mechanism' breaks down. The moisture content at which occurs may be referred to as the critical moisture content (Salomone et al., 1984). Similarly, during experiments regarding the impact of direct drying on FS, Makununika (2016) observed that the specific heat capacity and thermal conductivity of sludge pellets decreased with decreasing moisture content. Similar observations were made by Septien, Mugari *et al* (2018).

In general, at any given moisture content, a linear relationship exists between the thermal conductivity of a soil and dry bulk density or porosity (Farouki, 1986). At constant moisture content, as dry bulk density increases, the total void volume is reduced and connectivity between soil particles is increased resulting in increased thermal conductivity (Salomone and Kovacs, 1984; Kersten, 1949). However, at moisture contents near the critical moisture content, the effect of dry bulk density becomes minimal (Salomone and Kovacs, 1984). The thermal properties of the soil solids themselves also have an effect. For a fully saturated soil it has been found that an increase in dry bulk density results in the water in between pores being replaced with solids and the thermal conductivity may either increase – if the solid has a higher thermal conductivity than the water it replaces – or decreases with drying (Farouki, 1986). Dry bulk density has not been found to affect specific heat capacity but has been observed to affect the volumetric heat capacity at constant moisture content (Kersten, 1949; Boguslaw and Lukasz, 2003).

At a given moisture content and dry density, organic soils such as peat, and fine grained soils exhibit higher resistivity – lower conductivity – than coarse grained soils (Kersten, 1949; Salomone and Kovacs, 1984). This is because the thermal conductivity of organic material is relatively low as shown in Table 2-4 (Abu-Hamdeh and Reeder, 2000; Dong et al., 2015; Dissanayaka et al., 2013). Organic content has also been found to affect the specific heat capacity; the higher the organics proportion of a soil the higher the heat capacity as shown in Table 4-3 (Hanson et al., 2000). Therefore, it is likely that sludges exhibit relatively high specific heat capacities and low thermal conductivities if they are high in organics such as fresh FS and vice versa for sludges which are lower in organics such as stabilised FS.

Table 2-4. Approximate values of density and thermal properties of soil components at 10°C (de Vries, 1963; Hanson et al., 2000; Hillel, 1998)

Material	Density ρ (kg m^{-3})	Specific Heat Capacity c (kJ kg^{-1} °C^{-1})	Thermal Conductivity λ (W m^{-1} °K^{-1})
Quartz	2660.00	0.755	8.8
Clay minerals	2650.00	0.758	2.9
Soil organic matter	1275.00	1.932	0.25
Water	981.00	4.186	0.57
Air	1.20	1.005	0.025

It is important to note that the thermal conductivity of a given material with a given moisture content is not unique given its dependence on variable boundary conditions which may cause a moisture redistribution (Farouki, 1986). Therefore any measurements of thermal conductivity should be taken as implying effective rather than actual thermal conductivity (Farouki, 1986). From the previous discussion, specific heat capacity or volumetric heat capacity appear to be more robust measurements (Boguslaw and Lukasz, 2003). However, both measures are still dependent on other parameters. Essentially, the thermal conductivity of a soil depends on the volumetric properties of the soil components and the interfacial contacts between the soil and liquid phase (Dissanayaka et al., 2013). Whereas the soil heat capacity is controlled by the relative proportions of the soil components and their individual heat capacities (Dissanayaka et al., 2013).

2.3.5 Operational Parameters Which Affect the Drying of Faecal Sludge

To determine the impact of direct drying on sludge properties, Makununika (2016) characterised sludge for: moisture content; ash content; calorific value; thermal conductivity; specific heat capacity; elemental and molecular nutrients including magnesium, potassium, calcium, total phosphorous and nitrates. None of these properties were tested to determine the impact they have on drying, but rather the impact that drying has on them. Makununika (2016) found that both the thermal conductivity and specific heat capacity of the sludge increased with moisture content. Results from drying kinetics showed that air temperature, humidity and pellet diameter had a noticeable effect on drying time whilst the effect of air velocity was limited (Makununika, 2016). The drying rate curves reported showed a constant rate stage and a falling rate stage and it was noted that increasing the temperature affected the transition point i.e. the critical moisture content, between the two drying stages. An increase in temperature appeared prolonged the constant rate period until the sludge was much drier. Increased air humidity resulted in increased drying time of the sludge. It was noted that increasing the pellet diameter had a minor effect on the constant rate drying stage given it is predominantly a convective mass transfer process controlled by external

conditions. As the drying boundary layer occurred at the surface of the sludge pellet, an increase in diameter would have little effect on the rate of moisture evaporation from the pellet surface. However, the increase in diameter had a significant impact during the falling rate stage of drying. Results showed an increase in pellet diameter led to a longer falling rate stage due to the greater distance necessary for moisture to migrate to the surface.

Septien, Singh et al (2018) investigated latrine dehydration and pasteurisation (LaDePa) as a method the treatment of FS with radiative energy. In LaDePa ingoing FS is extruded into pellets and then exposed to infrared radiation. As the incoming sludge is extruded into pellets, sludge was characterised for rheological and plastic properties: viscosity and viscoelasticity were measured by rheometer, plasticity and the Atterberg limits were determined using a cone penetrometer. Further characterisation tests included moisture content, volatile solids and ash content – to determine thermal degradation – and nutrient content in terms of phosphorous, potassium, nitrogen and carbon to evaluate the usefulness of the end product in agriculture (Septien, Singh, et al., 2018). The calorific value was also reported to evaluate the potential of the sludge pellets as a biofuel. Drying curves plotted for different medium wave infrared (MIR) intensities showed, as expected, that an increase in MIR increased drying rate due to the increased temperature. Drying rate periods could not be identified as, although the drying rate decreased with time implying only a falling rate stage, the decrease may also have been due to pellet shrinkage and a reduced surface area. Thus, no conclusion could be drawn. Ash content and volatile solids showed no specific trend with varying MIR intensity implying little change in the "dry bone of the sludge" (Septien, Singh, et al., 2018). However, it should be noted once again that the properties reported were used to determine extent and effect of drying rather than if any of the properties affected the drying behaviour.

Septien, Mugari et al (2018) tested sludge samples using a solar drying rig. The sludge samples were placed inside the transparent drying chamber to determine the effect of the external weather conditions, air velocity and temperature and sample size. In sunny weather drying was faster when compared to overcast conditions and the transparent box acted as a greenhouse, improving drying rate of sludge in comparison to the sludge samples dried in the open air. Increasing the sample thickness led to an increase in drying time; however, increased diameter had little effect. This was assumed due to the fact that the mass per unit area to be dried did not change even with an increase in diameter: sample mass and surface area increased by the same amount (Septien et al., 2017). Temperature at the surface and in the core were not found to be significantly different in any sample so drying was considered isothermal. The reflectivity of the sludge samples was observed to decrease; the surfaces of

the sludge samples were observed to have a 'sheen' at the beginning of the drying tests but upon completion of the test, the surfaces were dull. The decrease in reflectivity would be expected to increase the absorption of solar radiation and therefore increase the rate of evaporation. Along similar lines, it is likely that the colour of sludge will affect the rate of evaporation as it is well established that darker colours conduct heat better since they absorb greater energy.

Septien *et al* (2017) also observed that the density of the sludge samples decreased with drying. This observation is in keeping with that of Radford and Fenner (2013) who found that the bulk density of pit latrine sludges was strongly negatively correlated to moisture content. However, Radford *et al* (2015) found no overall correlation between bulk density and water content. The lack of correlation noted by Radford *et al* (2015) was attributed to the variability in sludge samples which may have been caused by varying proportions of low-density fat and grease, and high-density sand and grit. It is unlikely that the density will affect the drying rate of FS but, based on soils theory, it may influence its thermal properties.

2.4 Modelling Sludge Drying

Drying curves may be regarded as a lumped parameter drying kinetics[10] model and have been found to adequately describe the drying process and give an accurate determination of drying time, provided the fit to the experimental data is reasonable. As the name suggests lumped parameter models require fewer equations and the medium to be dried is characterised by a few 'lumped' parameters (Kemp and Oakley, 2002). According to Kemp (2011) such lumped parameter models are "often easier to develop and more reliable in practice, giving useful insights for process understanding and control." However, the models are limited in that they cannot be theoretically predicted but must be measured and extrapolation to new conditions must be treated with caution (Kemp and Oakley, 2002). However, they may provide a useful starting point for the development of a model for the prediction of the required drying time for FS to reach a particular moisture content on a drying bed. Drying curves allow the estimation of the equilibrium moisture content which is used for thin-layer modelling of the drying of sludge.

[10] Vaxelaire and Puiggali (2002) define the drying kinetic as an overall approach to describe the behaviour of a medium during drying.

2.4.1 Equilibrium Moisture Content

The equilibrium moisture content is the moisture content at which a substance reaches equilibrium with the surrounding atmosphere and is determined by the temperature, relative humidity and properties of the substance itself (Hall, 1980; Peralta and Bangi, 2005; Vaxelaire and Cezac, 2004; Smollen, 1990). Thus, is cannot be said to be an intrinsic property of the substance being dried. In terms of drying, the equilibrium moisture content is important as, once it has been reached, no further drying of the substance is possible under those conditions (Spence et al., 2018). It can be found experimentally during drying tests as the moisture content of the substance at the point at which no further change in mass occurs (Basunia and Abe, 1999; Falade and Abbo, 2007). This is considered to occur when the change in mass of the sample is less than 0.05% of the total mass over a period of 24 hours in accordance with BS EN 12429 (1998). Sorption isotherm relations such as the GAB (Guggenheim-Anderson-DeBoer) equation are often used to estimate equilibrium moisture content and have been established and used successfully for timber (Peralta and Bangi, 2005; Simpson, 1973; Espinoza et al., 2007; P. R. Armstrong et al., 2012), as well as for abattoir paunch waste (Spence et al., 2018) and foodstuffs like figs (Xanthopoulos et al., 2007). The equilibrium moisture content is useful for calculating the moisture ratio which is the basis of many drying models.

The non-dimensional moisture ratio may be used to plot experimental data obtained from drying tests instead of moisture content (Fumagalli and Freire, 2007). Using moisture ratio instead of moisture content is useful as it allows easier comparison between samples of slightly differing initial moisture contents. The moisture ratio is a key component in the use of drying models to describe the drying kinetics of a medium. It can be calculated in terms of moisture content from:

$$MR_X = \frac{X - X_e}{X_0 - X_e}, \qquad (2.3a)$$

where MR_X is the dimensionless moisture ratio, X (kg kg^{-1} dry basis) is the moisture content at any instant, X_e (kg kg^{-1} dry basis) is the equilibrium moisture content and X_0 (kg kg^{-1} dry basis) is the initial moisture content (Lemus-Mondaca et al., 2009; Efremov and Kudra, 2004). Assuming that mass loss is due only to loss of moisture, MR may also be calculated in terms of mass[11] from:

[11] Where MR is expressed on a mass basis it may be more correctly termed mass ratio.

$$MR_M = \frac{M - M_e}{M_0 - M_e}, \tag{2.3b}$$

where MR_M is the dimensionless mass ratio, M (kg) is the sample mass at any instant, M_e (kg) is the mass at equilibrium moisture content, M_0 (kg) is the initial mass of the sample (Yaldýz and Ertekýn, 2001; Hossain and Bala, 2002; Fumagalli and Freire, 2007; Ademiluyi and Abowei, 2013; Spence et al., 2018). According to Yaldýz and Ertekýn (2001) and Midilli *et al* (2002), if the relative humidity of the drying air fluctuates, MR_X and MR_M may be simplified to:

$$MR_X = \frac{X}{X_0}, \tag{2.4a}$$

$$MR_M = \frac{M}{M_0}, \tag{2.4b}$$

where all terms are as defined previously. This simplification is based on the influence of relative humidity in determining equilibrium moisture content.

The moisture ratio and mass ratio are useful to allow comparison of the drying kinetics for products at different initial moisture contents (Vaxelaire and Puiggali, 2002). They are also often used as the dependent variable in thin-layer drying models (Lemus-Mondaca et al., 2009). Such models are commonly used to describe the drying kinetics of various foodstuffs such as apricots (Togrul and Pehlivan, 2002), roselle (Saeed et al., 2008), various vegetables (Yaldýz and Ertekýn, 2001) or popcorn (White et al., 1981). But they have also been used to successfully model the microwave drying of Zhaotong Lignite (Zhao et al., 2019), the convective drying of wastewater sludge (Cai et al., 2015), of sludge from treatment plants in tomato processing industries (Celma et al., 2012) and of real FS (Makununika, 2016). The following section provides a discussion of the models recommended for the analysis of the FS simulants dried during the drying test with UV lamp.

2.4.2 Thin Layer Drying Models

According to Kucuck *et al* (2014) thin-layer drying models fall predominantly into three categories: theoretical, semi-empirical (or semi-theoretical) and empirical. Semi-empirical models tend to be easier to use and require fewer assumptions due to the use of experimental data and are generally derived from either Fick's Second Law of Diffusion or are analogous to Newton's Law of Cooling (Kucuk et al., 2014). Newton's Law of Cooling assumes that the drying rate, *DR*, is directly proportional to the difference between actual and equilibrium moisture contents (Basunia and Abe, 1999; Kucuk et al., 2014). Assuming that change in mass is only due to loss of moisture and the thickness of the material to be dried is small enough

that evaporation of moisture from the surface is equal to the diffusion of moisture from the interior, then Newton's Law of Cooling may be written as:

$$drying\ rate, DR, \qquad \frac{dX}{dt} = -k(X - X_e), \tag{2.5}$$

where t (s) is drying time, k is the drying constant and all other terms are as defined previously. If the drying constant, k, is independent of X (kg kg^{-1} dry basis) and X_e, (kg kg^{-1} dry basis), then Equation 4.33 may be integrated to the single exponential equation:

$$MR = \exp(-kt), \tag{2.6}$$

where MR is as defined by Equation 4.31a or 4.31b or 4.32a or 4.32b and all terms are as defined previously (Hossain and Bala, 2002).

The Newton model is the simplest thin-layer drying model relating MR to time (Hossain and Bala, 2002; Saeed et al., 2008; Kucuk et al., 2014). It is widely used as a starting point for modelling of drying kinetics (Bruce, 1985). In deriving the Newton model, Lewis (1921) assumed constant drying conditions, limited shrinkage of the material and that drying occurred only in the falling rate period i.e. from the critical moisture content, W_{cl}. Table 2-5 shows some of the other semi-empirical thin-layer drying models commonly used to describe kinetics of the drying process which have built on the Newton model (Celma et al., 2012). These models will be used for the analysis of results from the drying of FS simulants with a UV lamp.

Table 2-5. Semi-empirical thin-layer drying models where MR is moisture ratio or mass ratio, k, k_0, k_1 are drying constants, t (s) is time and a, b and c are empirical constants and n is the drying exponent (adapted from Celma et al (2012))

Model	Equation		Reference
Newton	$MR = exp(-kt)$	(2.7)	Lewis (1921)
Page	$MR = exp(-kt^n)$	(2.8)	Page (1949)
Henderson and Pabis	$MR = a\ exp(-kt)$	(2.9)	Henderson and Pabis (1961)
Cai	$MR = a\ exp(-(kt)^n) + b$	(2.10)	Cai et al (2015)
Midilli-Kucuk	$MR = a\ exp(-kt^n) + bt$	(2.11)	Midilli et al (2002)
Logarithmic	$MR = a\ exp(-kt) + c$	(2.12)	Togrul and Pehlivan (2002)
Two term	$MR = a\ exp(-k_0 t) + \mathrm{b}\ \exp(-k_1 t)$	(2.13)	Henderson (1974)

A key limitation to the application of thin-layer semi-empirical drying models is that they are based on curve fitting of the experimental data with little to no understanding of the phenomena involved (Makununika, 2016). However, due to the ease of use and ability to adequately describe the drying kinetics of a medium, they are widely used in practice (Midilli,

H Kucuk, et al., 2002). Another limitation in the application of these models, specifically with regard to the drying of wastewater sludge is that they should only be applied for the falling rate period of drying (Kemp, 2011). As mentioned in Chapter 2, Section 2.2 , a constant rate and falling rate period, or periods are generally observed during drying of sludge, thus other another model is required to fully describe the drying kinetics.

2.4.3 Wastewater Sludge Drying Models

According to Kemp and Oakley (2002) scale-up from experimental design curves (e.g. drying kinetics) may often yield better results in practice than highly complex models which require a number of parameters. The experimental results presented from drying tests in Chapter 7 are in the form of drying curves and for this reason the remaining discussion will centre on models developed to describe the curves. Kemp (2011) recommended the use of simple first order kinetics to fit experimental data during the falling rate period of drying or a standard drying model which includes constant rate and falling rate periods. First-order kinetics is of special interest as it can be easily solved analytically, provided that values for the critical moisture content and equilibrium moisture content are known (Kemp and Sohet, 2010). For an industrial batch dryer, Kemp and Sohet (2010) recommended the following equation "to convert drying time from constant rate to a two-stage drying process with constant rate drying down to X_{cr} and first-order falling rate drying":

$$\frac{t_{2S}}{t_{CR}} = \left(\frac{X_i - X_{cr}}{X_i - X_O}\right) + \left(\frac{X_{cr} - X_e}{X_i - X_O}\right) ln\left(\frac{X_{cr} - X_e}{X_O - X_e}\right), \tag{2.14}$$

where t_{2s} (s) is two stage drying time i.e. total drying time, t_{CR} (s) is time for constant rate drying, X_i (kg kg^{-1} dry basis) is initial moisture content, X_{cr} (kg kg^{-1} dry basis) is critical moisture content, X_O (kg kg^{-1} dry basis) is moisture content at outlet and X_e (kg kg^{-1} dry basis) is equilibrium moisture content. For the analysis of drying kinetics of grass seeds Fumagali and Freire (2007) fit drying rate versus time curves and drying rate versus moisture content curves (Kirscher curves) to the following first-order kinetics and sigmoidal function respectively:

$$\frac{dX}{dt} = y_0 + a \cdot \exp(-bt), \tag{2.15}$$

$$\frac{dX}{dt} = y_0 + \frac{a}{1 + \exp[(x_0 - X)/b]}, \tag{2.16}$$

where y_0, a, b and x_0 are model constants and all other terms are as defined previously. The primary limitations of Equations 2.15 and 2.16 are that they are based on curve fitting, much like the previously discussed semi-empirical thin-layer drying models.

The two-stage model developed by Efremov (1998) for the convective drying of fibrous materials has been used to successfully simulate the drying kinetics of raw hog manure (Marzouk Benali and Kudra, 2002) and wastewater sludge (Reyes et al., 2004). The model was obtained by solving a one-dimensional isotropic diffusion equation (Efremov, 2000). For $t_r/\sigma \geq 2.7$ for drying a thin layer of material then:

$$\frac{X}{X_0} = \left(1 - N_0 \frac{t}{X_0}\right) + \frac{N_0\sqrt{\pi}}{2X_0}\left(1 - erf\left(\frac{t_f - t}{\sigma}\right)\right) = \left(1 - N_0 \frac{t}{X_0}\right) + \frac{N_0\sqrt{\pi}}{2X_0}\left(1 - erf(E)\right), \quad (2.17)$$

where N_0 is the initial drying rate (dX/dt), t_f (s) is final time, σ (s) is the characteristic time, E is dimensionless process time and all other terms are as defined previously. In Equation 2.17, *erf(E)* is an error function used in mathematical statistics and determined by the integral:

$$\mathrm{erf}(E) = \frac{2}{\sqrt{\pi}} \int_0^E e^{-t^2} dt\,, \quad (2.18)$$

where all terms are as defined previously (Efremov, 2000). The first term in Equation 2.17 represents the rate of moisture removal during the constant rate drying period (Marzouk Benali and Kudra, 2002; Efremov, 1998). The second term represents the falling rate period and "reflects the macroscale nonstationary diffusive moisture transport in a drying material, obtained by applying the Laplace transform method to the equation of isotropic diffusion with boundary conditions in the form of a constant concentration on the material surface" (Marzouk Benali and Kudra, 2002; Efremov, 2013). Assuming that equilibrium moisture content is obtained at the end of drying then Equation 4.57 becomes:

$$\frac{X_e}{X_0} = \left(1 - N_0 \frac{t}{X_0}\right) + \frac{N_0\sqrt{\pi}}{2X_0}, \quad (2.19)$$

and the characteristic drying time, σ (s), can be found from:

$$\sigma = \frac{2}{\sqrt{\pi}}\left(t_f - \frac{X_0 - X_e}{N_0}\right), \quad (2.20)$$

where all terms are as defined previously and N_0 may be determined experimentally or using steady-state heat-mass transfer equations (Efremov, 1998; M Benali and Kudra, 2002; Efremov, 2013). The characteristic drying time is "constant for a given process and is a function of both the nature of the material and the concrete drying conditions (temperature, moisture content, drying agent rate, etc)" (Efremov, 2000). To allow comparison of the drying kinetics of different materials under different conditions Efremov (2000) recommended the following equation:

$$K = \frac{X - X_e}{N_0 \sigma} = E - \frac{\sqrt{\pi}}{2}\mathrm{erf}(E)\,, \tag{2.21}$$

where all terms are as defined previously. Equation 2.21 may be particularly useful for future work given that it should allow comparison of drying of real FS with synthetic FS under similar conditions.

An issue in the use of Efremov's (1998; 2000) models is the difficulty often associated with determining the initial drying rate N_0. Efremov (2013) proposed the following generalised kinetic equation for constant rate drying:

$$X = X_0 - \frac{2X_0 - X_1 - X_2}{4} \cdot \frac{t}{\delta_2} - \frac{X_1 - X_2}{4} \cdot \frac{T - T_0}{\delta_1} \cdot \frac{t}{\delta_2}, \tag{2.22}$$

where X_0 (kg kg $^{-1}$ dry basis) is the initial moisture content, X_1 (kg kg $^{-1}$ dry basis) is final moisture content at first drying temperature, X_2 (kg kg $^{-1}$ dry basis) is final moisture content at second drying temperature, t (s) is drying time, δ_1 and δ_2 are interval of temperature variations, T (°C) is temperature of drying air and T_0 (°C) is temperature at centre (Efremov, 2013).

2.5 Faecal Sludge Simulants

The literature review encompassed simulants developed for fresh faeces, wastewater sludge and FS. In total 39 papers were found, of those 39, thirteen reported FS simulants and of those thirteen, seven (Byrne, 2012; Radford, 2011; Radford et al., 2015; Ha et al., 1986; Zuma, 2013; Penn et al., 2018; PRG, 2014b; Onabanjo, Kolios, et al., 2016) had developed original simulants. Eleven papers reported fresh faeces simulants, of which six had developed original simulants (Colón et al., 2015; Fisher et al., 1998; Podichetty et al., 2014; Wignarajah et al., 2006; Kaba et al., 1990; Yeo and Welchel, 1994). The remaining papers detailed simulants developed for wastewater sludge. An overview of the fresh faecal simulants and wastewater simulants found in the literature review is provided in Table 2-6 For a more in-depth discussion of the fresh faecal simulants refer to Appendix C.

Table 2-6. Overview of simulants developed for fresh faeces and wastewater sludge where N/S is not specified

	Reference	Type of Simulant	Purpose for simulant	Originality
Fresh Faeces	Afolabi *et al* (2014)	Fresh faeces	To test of hydrothermal carbonisation	Wignarajah *et al* (2006)
	Colon *et al* (2015)	High-strength undiluted excreta	To demonstrate suitability of using simple unmixed anaerobic digesters for treatment of excreta.	Adapted Wignarajah *et al* (2006)
	Danso-Boateng *et al* (2013)	Fresh faeces	To test of hydrothermal carbonisation	Fisher et al (1998)
	Fisher *et al* (1998)	N/S	To investigate waste incineration for resource recovery	Adapted Kaba *et al* (1990)
	Kaba *et al* (1990)	Sewage	To investigate electrochemical incineration of waste	Original
	Miller *et al* (2015)	Fresh faeces	To test supercritical water oxidation for FS treatment	Used Colon *et al* (2015)
	Podichetty *et al* (2014)	Fresh faeces	Viscous heating of faeces to destroy pathogens	Original
	Serio *et al* (2016)	Fresh faeces	To test a torrefaction (mild pyrolysis) processing apprach to sterilise faeces and produce a stable, solid product to be stored or recycled.	Used Wignarajah *et al* (2006)
	Ward *et al* (2014)	Fresh faeces	To evaluate the feasibility of the use of human faeces-derived char as a solid fuel.	Used Wignarajah et al (2006)
	Wignarajah *et al* (2006)	Fresh faeces	To allow safe testing of waste processing technologies in space systems	Adapted Kaba *et al* (1990)
	Wignarajah *et al* (2008)	Fresh faeces	To test of a microwave enhanced freexe drying solid waste processor to remove water from waste	Adapted Wignarajah et al (2006)
	Yeo & Welchel (1994)	Fresh faeces	Development of personal care devices such as nappies, training pants and incontinence pads.	Original
Wasteawater Sludge	Aslan *et al* (2014)	Brewery wastewater sludge	To investigate impact of different membrane modules on membrane fouling behaviour in a laboratory-scale submerged anaerobic membrane bioreactor	Original
	Banihani & Field (2012)	Wastewater sludge	To assess the performance of a combined system up-flow anaerobic sludge blanket and aerobic treatment activated sludge for the removal of carbonaceous and nitrogenous contaminants	Original

Wastewater Sludge	Baudez *et al* (2007)	Wastewater sludge	To produce inorganic and organic suspensions to describe behaviour of a real sludge	Original
	Boeijie *et al* (1998)	Wastewater sludge	To be used in laboratory-scale sewage treatment simulantions and biodegradation tests	Original
	Bracklow *et al* (2007)	Wastewater sludge	To test function of membrane bioreactors for carbon, nitrogen and biological phosphorous removal	Used Nopens *et al* (2001)
	Brown *et al* (2011)	Wastewater sludge	To test influence of anoxic and anaerobic hydraulic retention time on biological nitrogen and phosphorous removal in a membrane bioreactor	Original
	Dursun *et al* (2004)	Waste activated sludge	To replicate properties of real waste activated sludge	Original
	Fei (2013)	Wastewater sludge	To test accumulation of poly-hydroxybutyrates in anaerobic/anoxic/oxic membrane bioreactor	Used Brown *et al* (2011)
	Nguyen *et al* (2007)	Activated sludge	To study flocculation behaviour and final properties of synthetic and activated sludge	Used Sanin and Vesiling (1996)
	Nopens *et al* (2001)	Wastewater sludge	Storage of synthetic sludge	Adapted Boejie *et al* (1998)
	Ormeci & Vesilind (2000)	Activated sludge	To replicate dewatering properties of activated sludge	Adapted Sanin and Vesilind (1996)
	Sanin & Vesilind (1996)	Activated sludge	To investigate bioflocculation behaviour of activated sludge	Original
	Sanin & Vesilind (1999)	Activated sludge	To reproduce physical properties of activated sludge	Adapted Sanin and Vesilind (1996)
	Terry *et al* (1978)	Anaerobically digested wastewater sludge	To measure ammonia volitisation from wastewater sludge applied to soils	Original
	Yeneneh *et al* (2013)	Anaerobically digested wastewater sludge	To investigate the effect of ultrasonic, microwave and combined microwave-ultrasonic treatment on biogas production, solids removal and dewaterability of anaerobically digested sludge	Original

Simulants developed for type B FS are of primary interest. This type of sludge may be described as sludge that is generally retained with flush and greywater and normally originates from septic tanks or pits where it's stored from one to several years. During storage the sludge undergoes biochemical stabilisation to a significant extent and can therefore be considered relatively low strength (Heinss et al., 1998). In order to estimate whether the real FS that a simulant is meant to replicate is type A or type B, the containment technology, source and degree of stabilisation as determined by the volatile solids to total solids ratio or the organics content in terms of chemical oxygen demand or volatile solids. The volatile solids to total solids ratio is commonly "used as an indicator of the relative amount of organic matter and the biochemical stability of FS" (Strande et al., 2014). The higher the organics content the less stabilised a sludge is assumed to be (Strande et al., 2014; Heinss et al., 1998).

The earliest FS simulant recipe found in the literature was that developed by Ha *et al* (1986). The simulant was used in the development of a mathematical model for the anaerobic digestion of a mixture of night soil and septic tank sludge (Ha et al., 1986). The recipe developed was used to replicate a mixture of night soil and septic tank sludge of 7:3 and included proteose peptone, beef extract, ammonium bicarbonate, ammonium sulphate and other inorganics in very small concentrations. No rationale was given for the inclusion of any of the ingredients and it was noted that "unlike the actual waste, which contained a high concentration of solids, the synthetic waste was completely soluble" (Ha et al., 1986). Given the aim was to develop a model for the anaerobic digestion of the aforementioned sludge mixture the emphasis appeared to have been on replicating the chemical characteristics of the sludge which would affect anaerobic digestion and in using the volatile suspended solids (VSS) content to determine the actual microbial population. The simulant was assumed to have been intended to replicate the properties of type A FS. Anaerobic digestion is often used as a treatment option for sludges with high chemical and biochemical oxygen demands as would be expected for type A FS. Ha *et al* (1986) noted that the septic tank sludge had "already been through one type of biological treatment process [so] it was expected that the organic removal rate by biological activity would decrease as the proportion of septic tank sludge increased." From this it was inferred that night soil was less stabilised. The night soil formed the greater proportion of the real sludge indicating the sludge which the simulant was to replicate was less stabilised i.e. type A. Furthermore, the properties of the influent sludge mixture which the simulant was to replicate were similar to those reported for type A FS. For example, the night soil and septic tank mixture had a chemical oxygen demand of 47,700mg L^{-1} which is within the range reported for type A FS (20,000mg L^{-1} – 50,000mg L^{-1}) as reported by Strauss and Montangero (2002) and Heinss *et al* (1998).

The next simulant that was identified in the literature was that developed by Radford in 2011, a gap of 25 years. Radford (2011) developed a synthetic sludge to represent some mechanical properties of real FS to "enable like-for-like comparisons of the performance of different pit emptying technologies (PETs)… [and] facilitate prototype development". A simple two component simulant of kaolin clay and compost was proposed. Radford (2011) concluded that the simulant could replicate the full range of shear strengths of FS found in pit latrine sludge simply by varying the moisture content. However, the simulant was found to have a significantly lower bulk density than the real FS to which they were compared. The mix of 70% kaolin clay and 30% compost had a bulk density of between 800kg m^{-3} and 1200kg m^{-3} whilst the bulk density for real FS originating from pit latrines used for comparison lay between 1027kg m^{-3} and 1989kg m^{-3} (Radford, 2011; Radford and Fenner, 2013). The real FS used for comparison originated from pit latrines in Botswana where it was common practice to add sand to cover faeces following defecation. Radford and Fenner (2013) noted that the high bulk density values reported for real FS may have been due this practice as another study (Koottatep et al., 2012) reported lower bulk densities of real FS in the range of 1092 kg m^{-3} – 1159kg m^{-3}.

Depending on the duration of containment, user practice regarding pit additives and whether or not water is used for anal cleansing, FS originating in pit latrines may be type A or type B. Therefore, it may be inferred that the simulant developed by Radford (2011) could be representative of both type A and type B FS in terms of shear strength and density for sludges originating from lined pits where sand is not used as an additive (Radford and Fenner, 2013). Although it replicated the density of high and low strength sludge, the simulant did not replicate other key properties of real FS such as the total solids or organic matter content. Therefore, it is unlikely to prove useful for the purposes of replicating dewatering and drying behaviour of real FS.

Byrne (2012) proposed replacing compost with maize in the simulant recipe developed by Radford (2011). This substitution was intended to address the difficulties in regulating the moisture content of the simulant due to the high variability of moisture content in compost (Radford 2011). Byrne (2012) recommended maize as it has a less variable water content than compost. A simulant of 70% clay, 30% maize proved suitable in representing the density and shear strengths found in pit latrine FS for investigations into sludge pumpability and fluidisation and reduced variability amongst samples (Byrne, 2012). The density of the maize based simulant was higher than that reported for the Radford (2011) simulant by approximately 100kg m^{-3}, as shown in Figure 2-8. Based on the containment technology of the real FS used for comparison it was assumed that the simulant developed by Byrne (2012)

could replicate the density and shear strengths of both type A and type B sludge as both may originate from pit latrines. No extra data was available for estimating the stabilisation of the real sludge to be replicated using the simulant. As the simulant only replicated mechanical properties of real FS it is unlikely to be suitable to replicate the dewatering and drying behaviour of real FS. Furthermore, it was noted that the simulant could not be left to consolidate for long periods of time as it began to rot when left. This would render it unsuitable for testing in covered, USDBs as the testing period could last for ten days (as in this work) or longer.

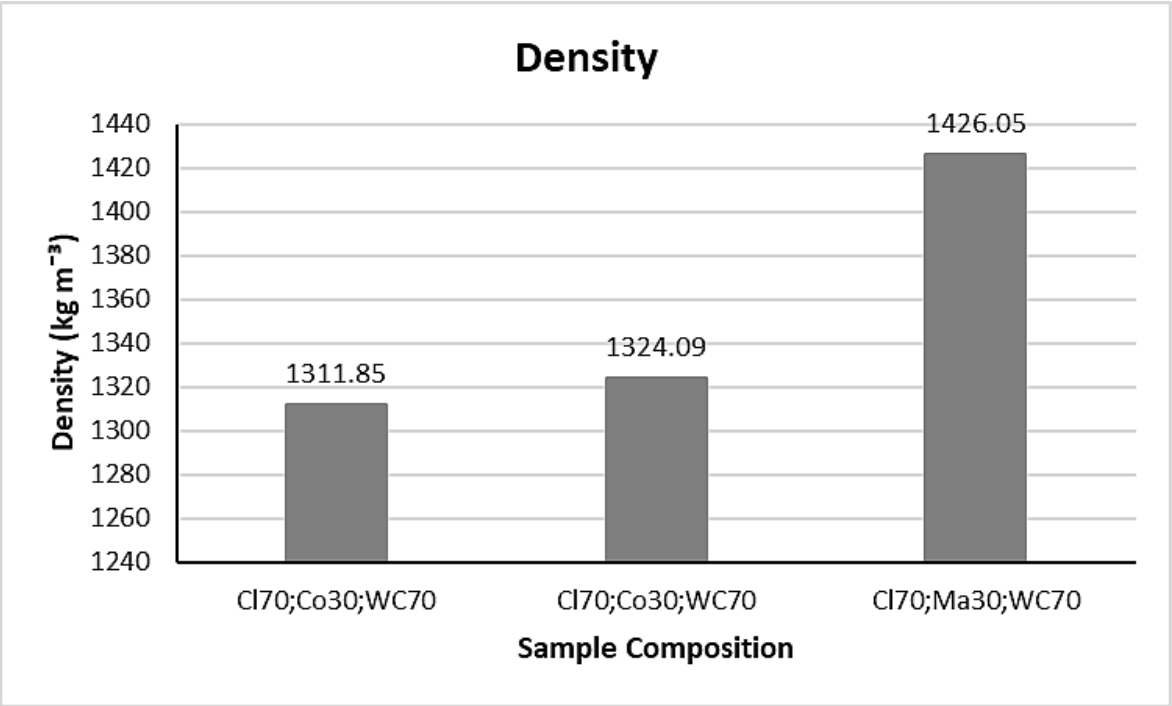


Figure 2-8. Density of compost based sample and maize based sample in same proportions where Cl is clay, Co is compost, WC is water content and Ma is maize (Byrne, 2012)

Radford *et al* (2015) built on the work of both Radford (2011) and Byrne (2012) to develop a simulant to replicate the mechanical properties of FS to "aid the development of desludging technologies." Radford *et al* (2015) recommended a simulant consisting of kaolin clay and topsoil or Milorganite, an organic fertiliser derived from sewage sludge, as suitable simulants. As was found with the Radford (2011) and Byrne (2012) simulants, the simulants recommended by Radford *et al* (2015) could replicate the full range of shear strengths reported for FS from different containment technologies including ventilated improved pit (VIP) latrines, urine-diversion (UD) toilets, school toilets, unimproved pits, community ablution blocks (CAB), pour-flush toilets and septic tanks. On the basis of bulk density measurements, Radford *et al* (2015) determined two broad groups of faecal sludge originating from various containment technologies. Group 1 had an average density of 1,100kg m^{-3} and consisted of sludge from unimproved pit latrines and septic tanks. Group 2 had an average

density of 1,400kg m^{-3} and included sludge originating from VIP latrines, UD toilets, school toilets and CABs. The density of the kaolin-topsoil simulant lay in the range of 1360kg m^{-3}-1450kg m^{-3} which was comparable with the range of densities reported for Group 2An average density of 980kg m^{-3} was reported for the Milorganite simulant which was comparable to the average density of FS from the containment technologies in Group 1. It is likely that the FS originating from Group 1 and Group 2 containment technologies are type B and type A respectively. Therefore, the kaolin-topsoil simulant replicated the density reported for type A sludges whilst the Milorganite simulant was comparable to the density of type B sludges. Neither simulant is likely to be useful in replicating the dewatering and drying behaviour of real FS as they are not representative of the chemical properties of real FS. Furthermore, Milorganite is difficult to obtain outside the USA (Radford et al., 2015).

Zuma (2013) developed a simulant to represent the FS characteristics that influence its biodegradation, including the main biological, physical and chemical characteristics. The simulant was also intended to represent the attachment sites for microorganisms on solid particles (Bassan et al., 2014). Hayflour was recommended as the main component of the simulant due to its wide availability and stable nature. It was also found to represent a number of properties of FS such as the total solids (TS), volatile solids (VS), suspended solids (SS) and volatile suspended solids (VSS). Following analysis, a mixture of hayflour and water was found to have a low chemical oxygen demand (COD) and lipid content so crushed walnuts were added to the recipe to increase these properties. Sodium phosphate (Na_2HPO_4) and ammonium carbonate (NH_4HCO_3) were added to represent the phosphorous and nitrogen content respectively. The simulant was varied as shown in Table 2-7 to represent different types of FS.

Table 2-7. Composition of components in different synthetic FS recipes where M (g) is mass, Na_2HPO_4 is sodium phosphate and NH_4HCO_3 is ammonium carbonate *(Zuma, 2013)*

Recipe	**1**	**2**	**3**	**4**	**5**
Component	**M (g)**	**M (g)**	**M (g)**	**M (g)**	**M (g)**
Walnuts	0.00	4.76	9.55	14.25	18.50
Hayflour	19.00	14.52	9.55	4.75	0.00
Na_2HPO_4	1.52	1.58	1.50	1.68	1.50
NH_4HCO_3	3.41	2.94	3.84	2.00	3.84

Each recipe was tested for the following properties: total solids; volatile solids; total suspended solids; total volatile solids; COD; soluble chemical oxygen demand (COD_s); total nitrogen; ammonium/ ammonia nitrogen; orthophosphate; and total phosphorous. According to Zuma (2013) these properties were chosen as they have previously been found to be "the most important parameters when it comes to sludge biological/chemical

processing technologies." Other properties which have been found to influence anaerobic digestion were also tested and included sulphate, proteins, carbohydrates, lipids, hemicellulose, cellulose and lignin (Zuma, 2013). Results from characterisation of the different simulant recipes showed that total solids appeared to have a significant impact on the values of other properties such as: total COD; COD_s; total protein and soluble protein; carbohydrates; and total sulphates and soluble sulphates.

The results of these tests were then compared with results from real FS from different countries and containment technologies i.e. type A and type B FS, to ascertain the most representative mixture. It was noted that no single recipe best represented the characteristics of FS from every containment technologies or country. Zuma (2013) compared the synthetic sludge with results reported by Koottatep *et al* (2001) for FS from septic tanks (septage) in Bangkok, as the total solids concentrations were similar. No single recipe was found to be most representative of the real FS. Zuma (2013) found that the average volatile solids of real FS was lower than all of the simulants as shown in Table 5-17. However, the volatile solids content reported for the simulants still fell within the range of 900 mg L^{-1} to 52,500 mg L^{-1} reported for real FS by Koottatep *et al* (2001). The suspended solids concentration of simulant recipe 3 (S3) matched most closely with that reported for real FS whilst total COD reported for simulant recipe 2 (S2) matched most closely as shown in Table 2-8. Furthermore, the volatile solids to total solids ratios for the simulant recipes shown in Table 2-8 was significantly higher than the average reported for real FS from Bangkok. The average ratio for real FS 71%, indicating a more stabilised sludge. This finding limits the applicability of this simulant to more stabilised sludge which has been found to have a volatile solids to total solids ratio of between 43% and 73% (Penn et al., 2018). The simulants developed by Zuma (2013) are instead more comparable to less stabilised, or type A FS.

Table 2-8. Average values of properties reported for simulants developed by Zuma (2013) and real faecal sludge (Koottatep et al., 2001) *where S is simulant, TS (mg L^{-1}) is total solids, VS (mg L^{-1}) is volatile solids, TSS (mg L^{-1}) is total suspended solids, VSS (mg L^{-1}) is volatile suspended solids*

Sludge		TS (mg L^{-1})	VS (mg L^{-1})	VS:TS	TSS (mg L^{-1})	COD (mg L^{-1})
S1[a]		18439.0	16380.0	0.88	14241.0	12470.0
S2[a]		17898.0	16568.0	0.93	17224.0	16667.0
S3[a]		17380.0	15882.0	0.91	14280.0	18467.0
S4[a]		19431.0	18584.0	0.96	16698.0	24500.0
S5[a]		19332.0	18420.0	0.95	17950.0	33483.0
Real FS[b]	Ave	19000.0	13500.0	0.71	15000.0	17000.0
	Range	2200.0 – 67200.0	900.0 – 52500.0	0.41 – 0.78	1000.0 – 44000.0	1200.0 – 76000.0

[a] Zuma (2013)

[b] Koottatep *et al* (2001)

The Pollution Research Group (PRG) at the University of KwaZulu-Natal (UKZN) developed a simulant to represent the chemical, biological and physical characteristics of FS as part of the Bill and Melinda Gates Foundation's Reinvent the Toilet Challenge (RTTC) Fair: India 2014. The simulant developed was based on simulants used for research at the University of Colorado Boulder and Duke University which were based on the recipes developed by Wignarajah *et al* (2006). The three 'base' recipes used by PRG are shown in Table 2-9. All three base recipes exchanged E.Coli for types of yeast and eliminate the dried coarsely ground vegetable matter from the recipes developed by Wignarajah *et al* (2006). Base recipe 1 appears to be similar to that proposed by Fisher *et al* (1998) with the addition of polyethylene glycol, PEG. Base recipe 2 is the same as that used by Miller *et al* (2015) and Colón *et al* (2015) discussed in the previous section.

Table 2-9. Simulant faecal sludge base recipes used by PRG (2014) where $\%M_{dry}$ is percent dry mass, PEG is polyethylene glycol, C_aPO_4 is calcium phosphate, NaCl is calcium chloride, KCl is potassium chloride and $CaCl_2$ is calcium chloride

Base Recipe #1 (PRG 2014)		**Base Recipe #2 (PRG 2014)**		**Base Recipe #3 (PRG 2014)**	
Component	**$\%M_{dry}$**	**Component**	**$\%M_{dry}$**	**Component**	**$\%M_{dry}$**
Nutritional yeast	30.0	Yeast[a]	30.0	Baker's yeast[a]	10.0
Cellulose	15.0	Cellulose	10.0	Cellulose	15.0
PEG	20.0			PEG	20.0
Psyllium	5.0	Psyllium	17.5	Psyllium	15.0
Peanut oil	20.0	Oleic acid	20.0	Peanut oil	5.0
Miso	5.0	Miso	17.5	Miso	30.0
$CaPO_4$	5.0	NaCl	2.0	$CaPO_4$	5.0
		KCl	2.0		
		$CaCl_2$	1.0		

[a] Yeast and baker's yeast are assumed to be analogous and refer to saccharomyces cerevisiae which is used to make bread products and can exist in the following forms: active dry; fresh; liquid; instant.

Each of the three recipes was altered in a variety of ways as shown in Table 2-10. These alterations were made to find a simulant which would best compare with real FS for a range of characteristics. Some substitutions such as for cellulose or calcium phosphate were made on the basis of readily available materials and cost.

Table 2-10. Substitutions and modifications for developing new simulants from base recipes shown in where PEG is polyethylene glycol and C_aPO_4 is calcium phosphate (adapted from PRG 2014)

Components	Substitution	Modification
Yeast	Fresh yeast; instant yeast	Reduced
Cellulose	Cotton balls and paper towels; shredded paper tissue; paper towels; sawdust; sawdust and tissue; cotton linters[a]; cotton linters and tissue	
PEG		Reduced
Psyllium		Reduced
Peanut oil		Reduced
Oleic acid	Olive oil	Reduced
Miso		Increased; reduced
$CaPO_4$	Prepared using eggshells as detailed in PRG (2014)	Reduced
Water		Added; increased; reduced

[a] Cotton linters are "the relatively short fuzz left on cottonseed after the cotton ginning process" (Orr, 1978). They are often used in the manufacture of paper and cellulose.

In total, ten alternative recipes were developed. All of the simulants, including the three base recipes were tested for the following properties: moisture content; total solids; ash; suspended solids; volatile solids; sludge volume index (SVI); COD; pH; density; thermal conductivity; specific heat capacity; calorific value; rheology; and particle size distribution (PRG, 2014b). Real FS was sampled from onsite sanitation facilities in the eThekwini Metro area around Durban, South Africa and characterised for the same properties to allow for comparison between each of the synthetic simulants and real FS. Results showed that synthetic simulant 9 (SS9) compared best with real FS in terms of chemical and physical characteristics (Velkushanova et al., 2017). A comparison of the results of various characterisation tests for real FS from various containment technologies and simulant SS9 is shown in Table 2-11.

Table 2-11. Summary of results of characterisation tests for faecal sludge from different containment technologies and simulant SS9 where OSS is onsite sanitation system, VIP is ventilated improved pit latrine, UD is urine diversion, CAB is community ablution block, Sch. is school, TS (%) is total solids, w (%) is moisture content, VS (g g dry sample^{-1}) is volatile solids, COD (g g dry sample^{-1}) is chemical oxygen demand, c (J kg^{-1} K^{-1}) is specific heat capacity, λ (W mK^{-1}) is thermal conductivity, ρ (kg m^{-3}) is density and all values reported are averages (PRG, 2014a)

Type of OSS	TS (%)	w (%)	VS (g g dry sample^{-1})	VS:TS	pH	COD (g g dry sample^{-1})	c (J kg^{-1}K^{-1})	λ (W mK^{-1})	ρ (kg m^{3})
Dry VIP	21.0	79.0	0.58	2.76	7.60	0.69	2530.54	0.54	1379.72
Wet VIP	21.0	79.0	0.54	2.57	7.59	0.69	2422.33	0.55	1447.78
UD toilet	43.0	60.0	0.45	1.04	7.54	0.49	2150.49	0.38	1450.37
CAB	23.0	77.0	0.49	2.13	7.44	0.65	3268.45	0.60	1350.10
Sch. Toilets	37.0	63.0	0.34	0.92	8.24	1.20	3509.58	0.55	1405.55
Unimproved pit	58.0	42.0	0.28	0.48	7.50	0.20	2366.89	0.29	921.45
SS9	19.0	81.0	0.75	3.95	5.91	1.04	3040.16	0.50	1300.00

As shown in Table 2-11 simulant SS9 (PRG, 2014b) had a similar total solids and moisture content to FS originating from dry and wet VIPs and community ablution blocks. However, the volatile solids and ash contents were much higher and lower respectively than for all of the real FS, likely due to the high proportion of organics in the simulant recipe. Similarly, COD was higher for the simulant than real FS from any containment technology except for school toilets. The specific heat capacity and density of the simulant were closest to the values reported for FS originating from a community ablution block. Thermal conductivity of FS originating from a dry VIP or wet VIP were the closest matches to the thermal conductivity of the simulant. Based on the volatile solids to total solids ratio, only the FS originating from unimproved pits was stabilised. The highest volatile solids to total solids ratio was calculated for FS originating from dry VIP latrines followed by wet VIP latrines indicating these sludges were the least stabilised. Therefore, the simulant sludge was most comparable to type A, un-stabilised FS.

The recipe for simulant SS9 (PRG 2014) is shown in Table 2-12. The simulant developed used approximately the same proportion of yeast as base recipes 1 and 2 but the percentage of cellulose and polyethylene glycol is lower than in all of the base recipes. The proportion of psyllium is higher than that in base recipe 1 and lower than in base recipes 2 and 3. Peanut oil is used in place of oleic acid, likely due to cost and availability, and is near to the proportion used in base recipe 1. The percentage of miso paste in the recommended simulant recipe is higher than that in base recipe 1 but lower than that in base recipes 2 and 3 whilst the proportion of calcium phosphate is double that in base recipes 1 and 3.

Table 2-12. Composition of synthetic FS where M (g) is mass, $\%M_{wet}$ is percent wet mass, $\%M_{dry}$ is percent dry mass, PEG is polyethylene glycol and C_aPO_4 is calcium phosphate *(PRG, 2014b)*

Components	**M (g)**	**$\%M_{wet}$**	**$\%M_{dry}$**
Instant yeast	72.8	7.3	32.49
Cellulose – half tissue and half cotton linters	12.4	1.2	5.53
PEG	27.2	2.7	12.14
Psyllium	24.3	2.4	10.84
Peanut oil	38.8	3.9	17.31
Miso	24.3	2.4	10.84
$CaPO_4$	24.3	2.4	10.84
Water	776.1	77.6	
TOTAL	**1000.2**	**100.0**	**100.0**

Onabanjo, Kolios *et al* (2016) and Onabanjo, Patchigolia *et al* (2016) adapted simulant SS9 (PRG 2014) to investigate the combustion performance of human faeces. However, propylene glycol was used in place of polyethylene glycol and cellulose powder was used in place of tissue and cotton linters. The simulant was successfully used for initial testing to "ensure repeatability, for fuel comparison and to understand the combustion operating conditions of the test rig" (Onabanjo, Kolios, et al. 2016).

Table 2-13 shows the average proximate and ultimate analyses of the feedstock for the experiments conducted, as well as the results reported by Velkushanova *et al* (2017) for the PRG (2014) simulant. Proximate (chemical) and ultimate analysis for synthetic faeces used by Onabanjo, Kolios *et al* (2016) and Onabanjo, Patchigolia *et al* (2016) and human faeces are similar; however, the synthetic faeces has a higher bulk density and lower average particle size than human faeces. The bulk density reported by Onabanjo, Kolios *et al* (2016) for the synthetic sludge is much lower in comparison to that reported by Velkushanova *et al* (2017). This may have been due to the methodology employed to determine density. However, no conclusion could be drawn in this regard as Onabanjo, Kolios *et al* (2016) did not specify the method used. The incomparability of results due to unspecified or different methodologies is a frequent issue in the area of FS due to a lack of standard operating procedures. The slight difference in the volatile solids content and ash content between SS (Onabanjo, Kolios, et al., 2016; Onabanjo, Patchigolla, et al., 2016) and SS9 (PRG, 2014b) may have been due to the differences in the two recipes, as the methodologies used to determine each property was the same.

Table 2-13. Proximate analysis and ultimate analysis of human faeces and simulants where M_{db} is mass on dry basis, VS is volatile solids, FC is fixed carbon, C is carbon, H is hydrogen, N is nitrogen, O is oxygen, HF is human faeces, SS is simulant faeces used by Onabanjo, Kolios et al *(2016) and Onabanjo, Patchigolia* et al *(2016), SS9 is PRG (2014) simulant and NR is not reported*

Sludge	Bulk density (kg m^{-3})	Particle size[a] (mm)	Proximate analysis ($\%M_{db}$)			Ultimate analysis ($\%M_{db}$)			
			VS	FC[b]	Ash	C	H	N	O[c]
HF[d]	277.0 ± 45.0	10.0 x 10	85.39	0.05	14.56	49.41 ± 0.11	7.62 ± 0.04	5.31 ± 0.02	23.10 ± 0.16
SS[a]	626.0 ± 5.0	5.0 x 10	86.77	0.08	13.15	44.85 ± 0.10	7.24 ± 0.03	2.52 ± 0.01	32.24 ± 0.10
SS9[e]	1300.0[f]	NR	75.00	NR	12.00	NR	NR	NR	NR

[a] Average size reported
[b] 100 – (%M of moisture, ash and VS) (Onabanjo, Kolios, et al., 2016)
[c] 100 – (%M of C, H, N and ash) (Onabanjo, Kolios, et al., 2016)
[d] Onabanjo *et al* (2016a)
[e] Velkushanova *et al* (2017)
[f] Dry bulk density reported

Penn *et al* (2018) developed a simulant based on the recipe developed by Colón *et al* (2015) which was based on the simulants developed by Wignarajah *et al* (2006). The purpose of the simulant was to replicate the physical, chemical and biological characteristics of real FS. Specifically, Penn *et al* (2018) aimed to develop a simulant which would replicate the movement, settling, sedimentation and dewatering, as well as the physical and biochemical disintegration, of excreta and FS in sewer pipes and onsite sanitation systems (OSS). The simulant recipe developed by Penn *et al* (2018) is shown in Table 2-14. The recipe used is distinct to that developed by PRG (2014) in two ways: yeast extract is added and may replace the baker's yeast component to replicate the microbial content and the inorganics used are the same as those used in the recipe developed by Colón *et al* (2015). Yeast extract was included after Penn *et al* (2018) noticed that the proportion of baker's yeast in the original simulant recipe – see Appendix B – created an "unfavourable physical structure [as] it produced a gassy and sticky material that floated when added to water, but was too sticky to be shaped into a cylinder" (Penn et al., 2018). The authors also noted that active yeast contributed to quick biological changes within the simulant which would be undesirable if the priority is reduced sample variability (Penn et al., 2018). In the event that a floating stool is required the developed simulant could include a small amount of baker's yeast as shown in Table 2-14 (simulants SB80 and SB65).

Table 2-14. Composition of synthetic faecal sludge simulants developed by Penn et al (2018) where TS is total solids, SB denotes simulant containing baker's yeast and yeast extract, SE denotes simulants containing only yeast extract, NaCl is sodium chloride, KCl is potassium chloride, $CaCl_2$ is calcium chloride and DI water is distilled water

Water content (%TS)	**80%**		**65%**	
Components	**SB80**	**SE80**	**SB65**	**SE65**
Yeast extract	65.06	72.29	105.42	126.51
Baker's yeast	7.23	0.00	21.08	0.00
Microcrystalline cellulose	24.10	24.10	42.17	42.17
Psyllium	42.17	42.17	73.80	73.80
Miso paste	42.17	42.17	73.80	73.80
Oleic acid	48.19	48.19	84.34	84.34
NaCl	4.82	4.82	8.43	8.43
KCl	4.82	4.82	8.43	8.43
$CaCl_2$	2.75	2.41	4.81	4.81
DI Water	758.7	758.7	577.72	577.72
Final mass (g)	**1000.00**	**1000.00**	**1000.00**	**1000.00**

Penn *et al* (2018) found that their simulant recipe could represent the range of moisture contents found in human faecal matter (65% to 80%). As shown in Table 5-24 the substitution of yeast extract for baker's yeast resulted in a simulant which is relatively similar in terms of chemical composition to real human faeces and exhibits similar physical characteristics, including shape formation. However, the viscosity of simulant SE80 is significantly higher than SB80 and is outside the range given for real human faeces as shown in Table 2-15.

Table 2-15. Comparison of chemical and physical properties of simulants SE80 and SB80 with real human faeces where SE denotes simulants containing only yeast extract, SB denotes simulant containing baker's yeast and yeast extract, COD_{total} (gCOD gTS^{-1}) is total chemical oxygen demand, COD_{sol} (gCOD gTS^{-1}) is soluble chemical oxygen demand, TN (%TS) is total nitrogen, EC is electrical conductivity, TS (%) is total solids and VS is volatile solids (Penn et al., 2018)

Property	SE80	SB80	HF
COD_{total} (gCOD gTS^{-1})	1.117 ± 0.056	1.194 ± 0.162	0.567 – 1.450
COD_{sol} (gCOD gTS^{-1})	0.624 ± 0.017	0.551 ± 0.048	NR
TN (%TS)	3.56 ± 0.13	4.05 ± 0.22	5.00 – 7.00
pH	5.4	5.2	4.6 – 8.4
EC	6.06 ± 0.17	6.40 ± 0.25	NR
TS (%)	20.56 ± 0.29	20.79 ± 0.30	14.00 – 37.00
VS (%TS)	87.61 ± 0.13	87.93 ± 0.07	92.00
Viscosity (cPs at 50rpm)	6360.00	4640.00	3500.00 – 5500.00
Density (g ml^{-1})	1.07 ± 0.02	0.98 ± 0.05	NR

Ward *et al* (2017) combined simulant SE80 with varying concentrations of the synthetic urine developed by Colón *et al* (2015) – see Table 5-13 – to investigate the dewaterability of FS. In total six types of FS were prepared to mimic high-strength to low-strength FS i.e. type A and type B sludge. Ward *et al* (2017) found that the simulants had a higher proportion of volatile solids than real FS. The authors attributed this to "the fact that synthetic FS did not undergo digestion, while FS from literature likely experienced degradation during storage in containment" (Ward et al., 2017). The simulant FS was found to have a 60% reduced dewaterability – as measured by centrifugation – compared to real fresh FS, which was likely due to the high water-binding affinity of the psyllium husk included in the simulant (Penn et al., 2018). However, Ward *et al* (2017) concluded that a simulant FS remained useful in the evaluation of the fundamental dewatering behaviour of real FS.

Research into FS simulants is in its early stages. There are several reasons why this is the case: FS is highly variable and there is still a lack of sufficient data on the properties of FS from different sources and containment technologies and how these change with time and stabilisation. This makes it difficult to develop a simulant which replicates the physical, chemical and biological properties of real FS. Thus, focus has tended to be on producing simulants which replicate certain properties which are relevant to the research being conducted, for example, Radford (2011). However, several simulants have been more recently developed to address these shortfalls and represent a range of chemical, physical and biological properties of real FS for example, PRG (2014) and Penn *et al* (2018).

2.6 Summary

In this chapter key properties, which affect the dewatering of FS by filtration and evaporation, were discussed and a summary is shown in Table 2-16. The type of sludge, property and effect

or likely effect are detailed. All of the properties found are dynamic and change with time. Additionally, they are often interlinked which adds significant complexity to the development of a representative simulant.

Table 2-16. Properties of wastewater sludge and faecal sludge that affect dewatering

Sludge Type	Property	Effect
Wastewater Sludge	Moisture Distribution	• Proportion of bound water will influence the rate and extent of filtration of a wastewater sludge assuming a high proportion of bound water will lead to low cake dry matter content and greater proportion of free water will result in increased rate of filtration. • Bound water content has been found to be affected by solids concentration, floc structure and strength and EPS content, particularly EPS protein content. • The distribution of water within a sludge affects the total drying time and length of drying periods assuming that the length of the constant rate period is determined by the amount of free water present.
	Total solids concentration	• Low total solids concentration results in low bound water concentration so that extent of filtration is increased.
	Organic matter content	• A high organics content leads to poor dewaterability.
	Particle size distribution	• Rate of dewatering is affected predominantly by the presence of supra-colloidal particles (1-100µm). • Presence of supra-colloidal particles is affected by sludge flocs which are affected by EPS content.
	Flocculation	• Primary determinant of settling and dewatering properties of wastewater sludge. • Flocs are significantly affected by EPS content.
	EPS	• A high proportion of EPS will result in high dewatering rate but low cake dry matter content and vice versa due to its water holding capacity, especially EPS protein content.
	Ionic composition and concentration	• The presence of cations has a significant effect on sludge flocculation. High concentrations of multivalent cations result in high cake dry matter content due to the formation of strong compact flocs but the presence of monovalent cations like sodium and magnesium leads to the breakdown of sludge flocs and release of fine particles resulting in low cake dry matter content. • pH affects the EPS content, at high pH the polymer network is destroyed, and fine particles are released resulting in poor filterability.
	Degree of Digestion	• A key factor in determining the ease with which sludge may drain as it impacts a number of sludge properties including particle size distribution, flocculation and EPS content.

Sludge Type	Property	Effect
Faecal Sludge	Moisture Distribution	• Proportion of free water will result in increased rate of filtration. • Affects the total drying time and length of drying periods assuming that the length of the constant rate period is determined by the amount of free water present.
	Moisture Content	• The total moisture content will affect the thermal properties of sludge.
	Total solids concentration	• A high initial total solids concentration has been found to results in lower rate of dewaterability but greater percentage of cake dry matter. • Solids concentration is affected by degree of stabilisation as well as the characteristics of the sludge capture and containment technology.
	Organic matter content	• Increased proportion of organics has been found to results in lower cake dry matter content, likely due to the water holding capacity of organics. • Composition of organic matter may be important as more stabilised sludge is likely to have a lower proportion of protein and therefore retain less water. • Organics content is affected by degree of stabilisation.
	Particle size distribution	• The greater percentage fines present the higher the CST value. • Particle size distribution is affected by degree of stabilisation and digestion, as well as containment technology and user practice.
	EPS	• Increase in EPS content, especially EPS humic content, correlated to an increase CST and turbidity. • Rather than contribute to flocculation, EPS is made of colloidal and suspended organic matter which contributes to turbidity and filter blinding. • EPS content in FS appears to be useful as a measure of the degree of stabilisation.
	Degree of stabilisation	• More stabilised sludge will dewater more quickly. • Influenced by the capture and containment technology, user habits, storage duration, inflow and infiltration, local climate and collection method.
	Ionic composition and concentration	• High proportion of monovalent cations, high conductivity and high pH all contributed to high CST values and greater turbidity which corresponds to poor sludge filterability.
	Reflectivity/Colour	• May have a limited impact on the rate of evaporation due to the effect on energy absorption.

Sludge Type	Property	Effect
Both	Specific heat capacity/ volumetric heat capacity	• Both are affected by water content, temperature and organic matter content. • Volumetric heat capacity is affected by dry bulk density.
	Thermal conductivity	• Thermal conductivity is affected by water content, dry bulk density, temperature, and organic matter content.
	Dry bulk density	• Dry bulk density affects thermal conductivity and volumetric heat capacity. • It is affected by the organic matter content.

There was a significantly larger volume of literature available regarding properties affecting the filterability of both wastewater and FS in comparison to drying. Most research regarding drying aimed to consider either the impact of external conditions on drying rates or on the properties of sludge following drying. Three key behaviours were highlighted which are often exhibited by sludge as it dries: cracking, shrinkage and crust formation. While shrinkage and crust formation are considered to retard the drying rate, cracking is considered to be beneficial and therefore encouraged. Shrinkage appears to be controlled by the presence or lack of a sludge "skeleton." Crust formation and cracking appear to be reliant upon external conditions and internal sludge properties although no literature was found detailing these properties.

Given the limited literature regarding the sludge properties affecting the drying rate of FS and the relevance to the drying tests in Chapters 5 and 6, external conditions affecting the drying rate of FS were discussed. Finally, a number of simulants developed for FS were discussed and it was highlighted that there is at present no simulant which replicates the physical and chemical properties of real, stabilised FS. In the following chapter, the development of faecal sludge simulants for characterisation and drying tests is discussed.

3 Faecal Sludge Simulant Development

In Section 2.4 of Chapter 2, a number of simulants were presented which have been developed for FS. Notably however, no simulant has yet been developed to represent the physical and chemical characteristics of stabilised FS which affect its dewatering behaviour. This chapter details the development of a number of simulants based on previously existing recipes developed for fresh FS.

3.1 Recommended Simulants for Further Development

The key properties found to significantly affect the dewatering of FS in the literature review were used to evaluate existing FS simulants to determine the most suitable for development as shown in Table 3-1. The FS simulants were also evaluated according to whether there was any mention of them exhibiting one or more of the key behaviours observed during drying discussed in Section 2.2 of Chapter 2. The first reference given (in bold) is for the original or adapted FS simulant recipe, any tests conducted by any other author using the same simulant is included in the same row. A simulant was considered to have been adapted if one or more ingredients was omitted or replaced. Where a square is blank it is because the property or behaviour was not reported for that particular simulant. In keeping with the definition given by Heinss *et al* (1998) type A FS is fresh while type B FS is stabilised.

There is a clear lack of FS simulants which are representative of type B FS in terms of all of the properties which were previously determined to affect the dewatering of real FS deposited onto USDBs. The simulants developed by Radford (2011), Byrne (2012) and Radford *et al* (2015) were noted previously as being able to replicate the mechanical properties of type A and type B FS by varying the moisture content. The remaining FS simulants (Ha et al., 1986; Onabanjo, Kolios, et al., 2016; Penn et al., 2018; PRG, 2014b; Zuma, 2013) were found to only be representative of fresh, type A FS. It was considered more desirable to adapt an existing simulant which represents the majority of the physical, chemical and thermal properties of real type A FS than to adapt the simulants which represented only the mechanical properties of type B FS.

The simulant developed by Ha *et al* (1998) reported no results for any of the key properties shown in Table 3-1and was previously noted as being completely soluble therefore it was discounted for further testing. The simulants developed by Onabanjo, Kolios *et al* (2016) and Zuma (2013) were found to be representative of limited properties affecting dewatering behaviour as shown in Table 3-1. The simulants which were found to be representative for the most properties of real type A FS were those developed by PRG (2014) and Penn *et al* (2018).

The simulant developed by PRG (2014) was the only simulant for which thermal properties were reported and these were comparable to real type A FS. Therefore, given the greater number of key properties which were reported for the simulants developed by PRG (2014) and Penn *et al* (2018), these two simulants were recommended for further development. There were no reports regarding cracking, crust formation and shrinkage for any simulant as none had been used for drying tests.

Table 3-1. Summary of the representativeness of key properties and behaviours exhibited during drying of simulants developed for faecal sludge where 0 indicates un-stabilised or fresh faecal sludge and 1 indicates stabilised faecal sludge

	Byrne (2012)	**Ha *et al* (1986)**	**Onabanjo, Kolios *et al* (2016);** Onabanjo, Patchigolia *et al* (2016)	**Penn *et al* (2018);** Ward *et al* (2017)	**PRG (2014)**	**Radford (2011);** Radford and Fenner (2013)	**Radford *et al* (2015)**	**Zuma (2013)**
Properties								
Moisture Distribution								
Moisture Content			x	x	x	x	x	
Total Solids Concentration				x	x			x
Organic Matter Content			x	x	x			x
Organic Matter Composition								
Particle Size Distribution			x		x			
Degree of Stabilisation	0/1	0	0	0	0	0/1	0/1	0
Ionic Composition and Concentration				x	x			
Reflectivity/ Colour				x	x			
Specific Heat Capacity/ Volumetric Heat Capacity					x			
Thermal Conductivity					x			
Density					x	x	x	
Behaviours								
Crust Formation								
Shrinkage								
Cracking								

To facilitate the development of new simulants, a list of ingredients was developed based on the FS simulants discussed in Chapter 2. The ingredients alongside the component(s) they are representative of and the supplier are shown in Table 3-2. The method for mixing and

storing FS simulants is provided in Section 3.2. Section 3.3 details the development of the PRG (2014) and Penn *et al* (2018) simulant recipes.

Table 3-2. Ingredients list for development of synthetic FS simulants where *PEG is polyethylene glycol*

Component	Molecular Formula	Organic/ Inorganic	Component representative of	Supplier	CAS Number
Instant yeast		Organic	Microbial content; carbohydrates; C	Sainsbury's	
Yeast extract	$C_{19}H_{14}O_2$	Organic	Microbial content; carbohydrates; C	Sigma Aldrich	0008013-01-2
Brewers yeast		Organic	Microbial content; carbohydrates; C	Buywholefoodsonline.co.uk	
Cotton Linters		Organic	Carbohydrates; cellulose	PULP - Paper Arts Workshop	
Shredded tissue		Organic	Carbohydrates; cellulose	Sainsburys	
Hemp fibre		Organic	Carbohydrates; cellulose	ACE Labs	
Microcrystalline cellulose	$C_{14}H_{26}O_{11}$	Organic	Carbohydrates; cellulose	Sigma Aldrich	0009004-34-6
Westland MPC (with added JI) compost		Organic	Water retention; cellulose; minerals	Homebase	
Psyllium husk	$C_{33}H_{66}O_2$	Organic	Carbohydrates (fibre)	Buywholefoodsonline.co.uk	
Regucol		Organic	Bacterial debris; carbohydrates (fibre)	Holland and Barrett	
Peanut oil		Organic	Fats and lipids	Amazon.co.uk	
Oleic acid	$C_{18}H_{34}O_2$	Organic	Fats and lipids	Sigma Aldrich	0000112-80-1
Miso paste		Organic	Proteins; fats; minerals; carbohydrates (fibre)	Yutaka Organic Miso Paste – Amazon.co.uk	
PEG E 400	$C_{16}H_{34}O_9$	Organic	Water retention	Sigma Aldrich	0025322-68-3
Cattybrook clay		Inorganic	Water retention; inorganics	ACE Labs	
Calcium phosphate	$Ca_3O_8P_2$	Inorganic	P	Fisher Scientific	
Sodium chloride	NaCl	Inorganic	Na	Sigma Aldrich	0007647-14-5
Potassium chloride	KCl	Inorganic	K	Sigma Aldrich	0007447-40-7
Calcium chloride	$CaCl_2$	Inorganic	Ca	Sigma Aldrich	0010035-04-8
Sodium phosphate	Na_2HPO_4	Inorganic	P		
Ammonium bicarbonate	NH_4HCO_3	Inorganic	Ammonia nitrogen	Sigma Aldrich	0001066-33-7

3.2 Methodology for Preparation and Storage of Faecal Sludge Simulants

All simulants were prepared according to the following method:

3. Weigh dry ingredients and add to beaker in the order listed on the recipe. Note that dry ingredients include any polyethylene glycol, miso paste, peanut oil or oleic acid.
4. Mix the ingredients with a small spatula until combined and the consistency of a thick paste.
5. Add the required amount of water.
6. Mix paste with water slowly with small spatula until an even consistency is obtained and there is no more paste at the bottom of the beaker.
7. Mix synthetic simulant with a hand blender for 30.0 s to ensure a homogenous consistency.
8. Store the mixed simulant in an airtight container in a refrigerator at 4°C until required.
9. Prior to analysis, remove the simulant from the refrigerator and allow to sit at room temperature for up to two hours. Prior to removal from container, mix simulant thoroughly for 1.0 minute with a small spatula to ensure homogeneity prior to sampling for analysis.

All recipes listed in this thesis provide 500g of simulant, should more or less be required the recipe may be scaled up or down according to the percentage dry mass also provided in the recipes.

3.3 Alterations to Faecal Sludge Simulant Recipes

The recipe used for the FS simulant developed by PRG (2014) is shown in Table 3-3 The simulant developed by Penn *et al* (2018) which contained only yeast extract and had a moisture content of 80% was used for development and is shown in Table 3-4. It is distinct from the recipe developed by PRG (2014) in that it requires microcrystalline cellulose and oleic acid for the cellulose and lipids components respectively. The recipe developed by Penn *et al* (2018) does not include polyethylene glycol and calls for sodium chloride, potassium chloride and calcium chloride to represent the minerals component in place of calcium phosphate. Furthermore, the proportions of most of the components in each recipe are different.

Table 3-3. Composition of faecal sludge simulant developed by PRG (2014) where M (g) is mass, $\%M_{wet}$ is percent wet mass, $\%M_{dry}$ is percent dry mass, PEG is polyethylene glycol and C_aPO_4 is calcium phosphate *(PRG, 2014b)*

Components	**M (g)**	**$\%M_{wet}$**	**$\%M_{dry}$**	**Component representative of**
Instant yeast	36.40	7.30	32.49	Microbial content
Cellulose – half tissue and half cotton linters	6.20	1.20	5.53	Cellulose/ carbohydrates
PEG 400	13.60	2.70	12.14	Water retention
Psyllium	12.15	2.40	10.84	Fibre; carbohydrate
Peanut oil	19.40	3.90	17.31	Fats and lipids
Miso	12.15	2.40	10.84	Fibre; protein; fat; nitrogen
$CaPO_4$	12.15	2.40	10.84	Inorganics
Water	388.05	77.60		Water
TOTAL	**500.10**	**100.00**	**100.00**	

Table 3-4. Composition of faecal sludge simulant SE80 developed by Penn et al (2018) where SE denotes a simulant containing only yeast extract, $\%M_{dry}$ is percent dry mass, NaCl is sodium chloride, KCl is potassium chloride, $CaCl_2$ is calcium chloride and DI water is distilled water

Components	**M (g)**	**$\%M_{dry}$**	**Component representative of**
Yeast extract	36.15	30.0	Microbial content
Microcrystalline cellulose	12.05	10.0	Cellulose/ carbohydrates
Psyllium	21.09	17.5	Fibre; carbohydrate
Miso paste	21.09	17.5	Fats and lipids
Oleic acid	24.10	20.0	Fibre; protein; fat; nitrogen
NaCl	2.41	2.0	Inorganics
KCl	2.41	2.0	
$CaCl_2$	1.21	2.0	
DI Water	379.35		Water
TOTAL	**499.48**	**100.0**	

Despite the differences between the two simulant recipes they had similar moisture contents and total solids content as shown in Table 3-5. The moisture content and total solids content of each FS simulant was representative of real from dry and wet VIP latrines. The recipe developed by Penn *et al* (2018) contained a higher volatile solids content than that developed by PRG. Although both recipes had higher volatile solids contents than real FS as shown in Table 6.3. This is likely due to the higher proportion of organic ingredients used in each recipe (Penn et al., 2018). The simulant developed by Penn *et al* (2018) appeared to have a lower density than that reported for real FS from different containment technologies and the PRG (2014) simulant. However, different methodologies were used to determine density: PRG (2014) determined density using a C-Therm TCi small volume test kit (SVTK) whilst Penn *et*

al (2018) determined density by displacement of water following the addition of a sample. Greater emphasis was placed on the development of the PRG (2014) simulant since it had previously been found representative of real FS in terms of thermal properties as shown in Table 3-5. It also had a lower volatile solids concentration and higher density than the simulant developed by Penn *et al* (2018) and was therefore more similar to real FS in these respects.

Table 3-5. Summary of results of characterisation tests for faecal sludge from different containment technologies and faecal sludge simulants developed by PRG (2014) and Penn et al (2018) where OSS is onsite sanitation system, VIP is ventilated improved pit latrine, UD is urine diversion, CAB is community ablution block, TS (%) is total solids, w (%) is moisture content, VS (g g dry sample^{-1}) is volatile solids, c (J kg^{-1} K^{-1}) is specific heat capacity, λ (W mK^{-1}) is thermal conductivity, ρ (kg m^{-3}) is density

Type of OSS	TS (%)	w (%)	VS (g g dry sample^{-1})	Ash (g g dry sample^{-1})	pH	c (J kg^{-1}K^{-1})	λ (WmK^{-1})	ρ (kg m^{3})
Dry VIP[a]	21.0	79.0	0.58	0.42	7.6	2530.54	0.54	1379.72
Wet VIP[a]	21.0	79.0	0.54	0.46	7.59	2422.33	0.55	1447.78
UD Toilet[a]	43.0	60.0	0.45	0.27	7.54	2150.49	0.38	1450.37
CAB[a]	23.0	77.0	0.49	0.51	7.44	3268.45	0.60	1350.10
PRG (2014)[a]	19.0	81.0	0.75	0.12	5.91	3040.16	0.50	1300.00
Penn *et al* (2018)[b]	20.56	80.0	0.88	NR	5.4	NR	NR	1070.00

[a] Velkushanova *et al* (2018)
[b] Penn *et al* (2018)

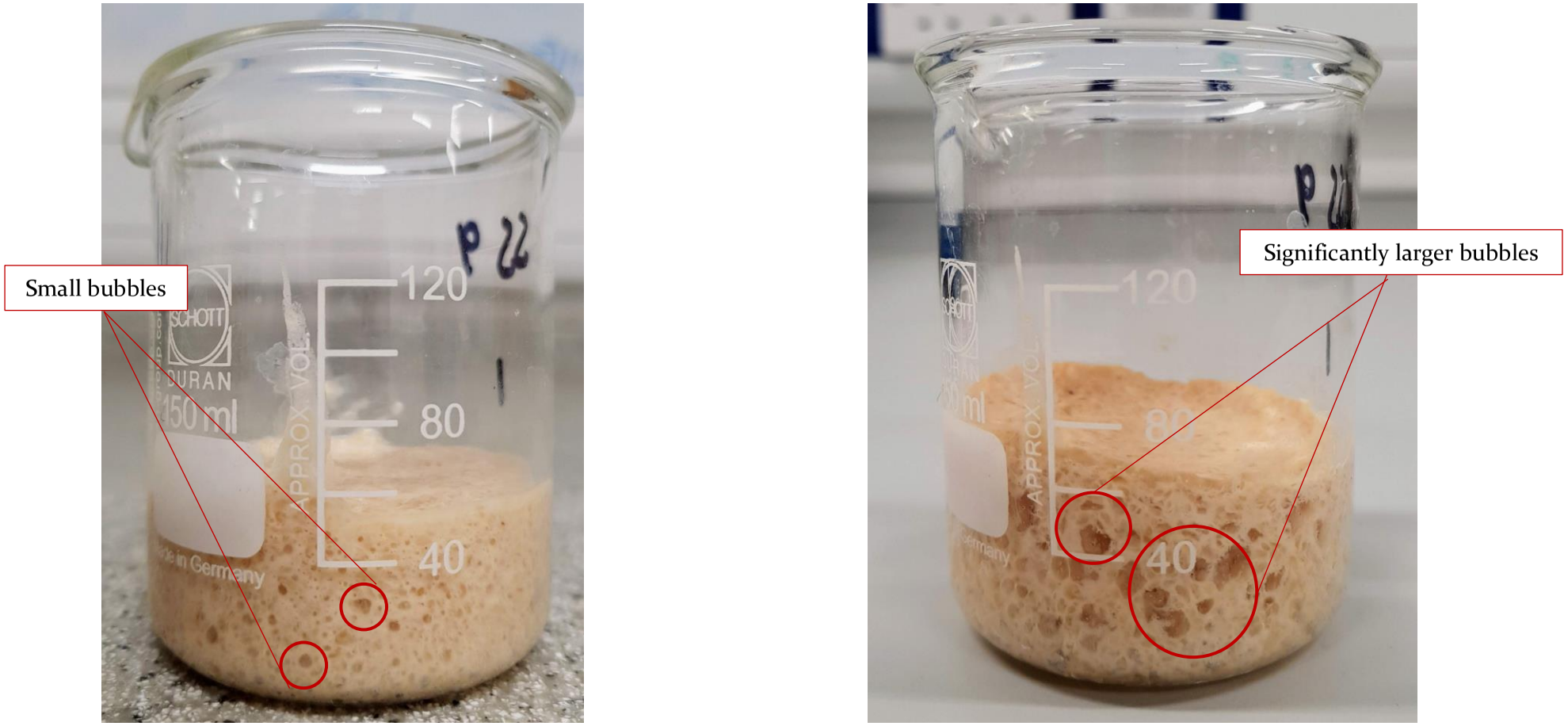
Based on availability of materials it was necessary to make several alterations to the original PRG (2014) simulant recipe given in Table 3-3. These initial alterations (SS9, BM1 and BM2) are shown in Table 3-6. Cotton linters were only available in pressed sheets instead of the more fibrous material used by PRG (2014) originally. Therefore, the first altered simulant, SS9, was made to ascertain whether the form of cotton linters affected the characterisation test results. Due to availability, PEG 3350 was used in place of PEG 400. Apart from these two substitutions, the components of SS9 were the same as those of the original PRG (2014) recipe provided in Table 3-3. On the basis that hemp fibre consists mainly of cellulose and hemi-cellulose, and assuming that the fibrous material would perhaps make a better substitute for the fibrous cotton linters in the original PRG (2014) recipe, it was used in Bath Mix 1 (BM1). Hemp fibre made up the entire cellulose component for Bath Mix 2 (BM2) to see what impact, if any, it would have. The added advantage of hemp fibre is its global and local availability. The proportions of each component for simulants SS9, BM1 and BM2 are

not shown in Table 3-6 as they remained the same as for the original PRG (2014) recipe shown in Table 3-3.

Table 3-6. Initial alterations made to the PRG (2014) faecal sludge simulant where SS is synthetic sludge, BM is Bath mix, PEG is polyethylene glycol and C_aPO_4 is calcium phosphate

SS9	BM1	BM2	Component representative of
Instant yeast	Instant yeast	Instant yeast	Microbial content
Half cotton linter (pressed sheet)/ half shredded tissue	Half shredded tissue/ half hemp fibre	Hemp fibre	Cellulose/ carbohydrates
PEG 3350	PEG 3350	PEG 3350	Water retention
Psyllium husk	Psyllium husk	Psyllium husk	Fibre; carbohydrate
Peanut oil	Peanut oil	Peanut oil	Fats and lipids
Miso paste	Miso paste	Miso paste	Fibre; protein; fat; nitrogen
$CaPO_4$	$CaPO_4$	$CaPO_4$	Inorganics
Water	Water	Water	Water

As expected, after mixing the first batches of each of the simulants (SS9, BM1 and BM2) detailed in Chapter 3, Section 3.2 it was noted that bubbles began to form. When samples of each simulant were left for a period of 24 hours, all samples became full of bubbles and rose due to the reaction of the yeast in the recipes as shown in Figure 3-1. This is in keeping with the observations made by Penn *et al* (2018).



(a) (b)

Figure 3-1. Photographs of PRG SS9 showing bubble formation: (a) at 0 hours (b) at 24 hours (photographed by author (2018))

To address the issue, four more recipes were developed which altered simulants BM1 and BM2 by replacing active yeast with inactive nutritional brewers yeast (Yesto-Seal® 004) or eliminating the yeast entirely. As shown in Table 3-7 Bath Mix 3 (BM3) was made using half hemp fibre and half shredded tissue for the cellulose component and brewers yeast. Bath Mix 4 (BM4) was also made with brewers yeast but with all hemp fibre as the cellulose component. Bath Mix 5 (BM5) and Bath Mix 6 (BM6) were made eliminating yeast entirely to ascertain the affect that this would have on the simulant characteristics. Eliminating the yeast also served to make certain whether it was the active yeast causing the simulants (SS9, BM1 and BM2) to bubble or if another reaction between the components of the recipes was responsible.

Table 3-7. Further alterations to faecal sludge simulants BM1 and BM2 where BM is Bath mix, PEG is polyethylene glycol and C_aPO_4 is calcium phosphate

BM3	BM4	BM5	BM6	Component representative of
Brewers Yeast	Brewers Yeast			Microbial content
Half shredded tissue/ half hemp fibre	Hemp fibre	Half shredded tissue/ half hemp fibre	Hemp fibre	Cellulose/ carbohydrates
PEG 3350	PEG 3350	PEG 3350	PEG 3350	Water retention
Psyllium husk	Psyllium husk	Psyllium husk	Psyllium husk	Fibre; carbohydrate
Peanut oil	Peanut oil	Peanut oil	Peanut oil	Fats and lipids
Miso paste	Miso paste	Miso paste	Miso paste	Fibre; protein; fat; nitrogen
$CaPO_4$	$CaPO_4$	$CaPO_4$	$CaPO_4$	Inorganics
Water	Water	Water	Water	Water

It was observed that the PRG SS9 and BM simulants were sticky in consistency and would likely exhibit poor solid-liquid separation as very little free moisture could be seen. Therefore, four more simulants were developed based on the ingredients list shown in Table 3-3 and on the proportions of ingredients from the PRG (2014), Penn *et al* (2018) and Wignarajah *et al* (2006) simulant recipes. The proportions of the components in simulant recipes Naomi Deering (ND) 1-4 are shown in Table 3-8.

Yeast extract was chosen to represent bacterial debris for ND1-3 given that it had been successfully used in eliminating bubble formation in the simulant developed by Penn *et al* (2018). Yeast extract was replaced with *Regucol* in simulant ND4. *Regucol* was chosen as it contained a combination of psyllium husk, inulin (a water-soluble fibre and natural carbohydrate) and the bacterial cultures lactobacillus acidophilus and bifidobacterium bifidum thus representing a number of components in the simulant recipe.

Westland Multi-Purpose Compost (MPC) with John Innes was included in all four recipes. John Innes (JI) are a "range of composts developed at the John Innes Institute" (Gardening Data, 2019). The type of JI used in the MPC is not specified but generally the composts consist of loam – which forms the base of the compost – peat, sand and fertiliser which contains nitrogen, phosphates and other nutrients useful for plant growth. According to the manufacturer Westland MPC with JI contains a high proportion of peat (approximately 70.0%) to aid water retention, has a neutral pH and contains nutrients such as nitrogen and phosphates for plant growth thus it serves multiple purposes in the simulant recipes including water retention and representing the cellulose component.

Peat is well known for its water retention ability (Andriesse, 1988). It consists of a very high proportion of organic matter made of varying constituents which can be divided into four groups based on their solubility in aqueous and organic solvents: bitumens; carbohydrates; lignin and lignin-like materials; and humic substances (Delicato, 1996). Carbohydrates make up the largest fraction of the organic matter in peat including carbohydrates such as cellulose and hemicellulose. Therefore, the compost should increase the cellulose component in the ND simulant recipes. This was considered desirable as increasing the proportion of cellulose was anticipated to result in simulants which were more representative of stabilised sludge. This is because stabilisation of real FS results in a sludge which contains "organic, carbon-based molecules that are not readily degradable, and... [consist] of more stable, complex molecules (e.g. cellulose and lignin)" (Strande et al., 2014). Additionally, compost was used previously in the FS simulant recipe developed by Radford (2011). Prior to adding to the mix, compost was dried in an oven at 105°C for 48 hours. This was to ensure that the high variability of the moisture content in compost would not cause the same difficulty in varying moisture content as was noted by Radford (2011). The bone-dry compost was passed through a 2mm sieve to ensure removal of grit and stones and then blended in a food processor for approximately two minutes.

A small amount of Cattybrook clay was to decrease the volatile solids content in simulants ND3 and ND4. PEG 3350 was eliminated from simulants ND3 and ND4 on the basis that the clay would also absorb some of the water added to the simulant thus representing water retention. The clay may also aid in the development of cracking during drying tests. According to Hillel (1998) clay particles adsorb molecules and hydrate causing the soil they are in to swell upon wetting and shrink and crack upon drying.

The proportion of psyllium husk was reduced simulants ND1, ND2 and ND3. It was thought that limiting the psyllium husk content would perhaps result in a simulant which did not retain as much water as psyllium husk swells on contact with water forming a mucilaginous

mass (National Centre for Biotechnology Information, n.d.). Penn *et al* (2018) noted that psyllium husk could be to blame for the reduced dewaterability of the simulants tested by Ward *et al* (2017) in comparison with real FS. No psyllium husk was included in simulant ND4 as *Regucol* contained psyllium. A small amount of hemp fibre was used in ND2 to see if it acted like a skeleton to prevent shrinkage. The proportions of inorganics in each simulant was varied.

Table 3-8. Recipes of simulants ND1, ND2, ND3 and ND4 developed at the University of Bath where $\%M_{dry}$ is percent dry mass, PEG is polyethylene glycol, KCL is potassium chloride, $CqCl_2$ is calcium chloride, $Ca_3O_8P_2$ is calcium phosphate, NH_4HCO_3 is ammonium bicarbonate and NaCl is sodium chloride

	ND1	**ND2**	**ND3**	**ND4**	
Component	**$\%M_{dry}$**	**$\%M_{dry}$**	**$\%M_{dry}$**	**$\%M_{dry}$**	**Component representative of**
Yeast extract	30.00	20.00	30.00		Microbial content; carbohydrates (fibre)
Regucol				30.00	Microbial content; carbohydrates (fibre)
Compost	20.00	20.00	20.00	20.00	Cellulose; fibre
PEG E 400	7.00	7.00			Water retention
Clay			6.50	8.50	Water retention; inorganics
Miso paste	20.00	20.00	20.50	20.50	Proteins; fats; minerals; carbohydrates (fibre)
Peanut oil	15.00		15.00	15.00	Fat and lipids
Oleic acid		25.00			Fat and lipids
Hemp fibre		2.00			Cellulose
KCl	2.00	1.00	2.00	2.00	Minerals
$CaCl_2$	1.00		1.00	1.00	Minerals
$Ca_3O_8P_2$		1.00	1.00	1.00	Minerals
NH_4HCO_3		1.00	2.00	2.00	Ammonia nitrogen
NaCl	2.00				Minerals
Psyllium husk	3.00	3.00	2.00		Carbohydrates (fibre)
Water					Water
TOTAL	**100.00**	**100.00**	**100.00**	**100.00**	

The proportion of water added to each recipe was calculated in the same manner as described by Wignarajah *et al* (2008). The dry ingredients (including the oleic acid or peanut oil and PEG) for each simulant were mixed and dried in an oven at 105°C for 24 hours to determine the inherent water content (Wignarajah et al., 2008). The inherent water contents for each simulant were determined in triplicate and the average value for each is reported in Table 3-9. In order to match the moisture content reported for real FS and the simulants developed

by PRG (2014) and Penn *et al* (2018), water was added to each ND simulant recipe to generate an approximate water content of 80%. The composition and final mass of the ND simulants including water is shown in Table 3-10.

Table 3-9. Inherent moisture content (w) of dry ingredients of simulants ND1, ND2, ND3 and ND4

Simulant	**w (%)**
ND1	16.46
ND2	16.86
ND3	18.14
ND4	14.90

Table 3-10. Composition of synthetic faecal sludge simulants ND1, ND2, ND3 and ND4 developed at the University of Bath where M (g) is mass, *PEG is polyethylene glycol, KCL is potassium chloride,* $CqCl_2$ *is calcium chloride,* $Ca_3O_8P_2$ *is calcium phosphate,* NH_4HCO_3 *is ammonium bicarbonate and* NaCl *is sodium chloride*

	ND1	**ND2**	**ND3**	**ND4**	
Component	**M (g)**	**M (g)**	**M (g)**	**M (g)**	**Component representative of**
Yeast extract	24.69	16.46	27.21		Microbial content; carbohydrates (fibre)
Regucol				22.35	Microbial content; carbohydrates (fibre)
Compost	16.46	16.46	18.14	14.90	Cellulose; fibre
PEG E 400	6.17	5.76			Water retention
Clay			5.90	6.33	Water retention; inorganics
Miso paste	16.87	16.46	18.59	15.27	Proteins; fats; minerals; carbohydrates (fibre)
Peanut oil	12.35		13.61	11.18	Fat and lipids
Oleic acid		20.58			Fat and lipids
Hemp fibre		1.65			Cellulose
KCl	1.65	0.82	1.81	1.49	Minerals
$CaCl_2$	0.82		0.91	0.75	Minerals
$Ca_3O_8P_2$		0.82	0.91	0.75	Minerals
NH_4HCO_3		0.82	1.81	1.49	Ammonia nitrogen
NaCl	1.65				Minerals
Psyllium husk	1.65	2.47	1.81		Carbohydrates (fibre)
Water	417.70	417.70	409.30	425.50	Water
TOTAL	**500.00**	**500.00**	**500.00**	**500.00**	

3.4 Summary

In this chapter previously developed FS simulants were assessed against the key properties and behaviours affecting the dewatering of real FS. No simulant was found to be representative of all of the key physical, chemical and thermal properties highlighted for real stabilised FS. It was deemed more suitable to adapt one or two simulants which represented a majority of the key properties than to adapt one of the simulants which was only representative of the mechanical properties of real FS i.e. The simulants developed by PRG (2014) and Penn *et al* (2018) were recommended development as they were found to be representative of real type A FS for the majority of the key properties identified. In total 11 new simulants were developed. Simulants PRG SS9 and BM1-6 were adaptations of the recipe developed by PRG (2014) and simulants ND1-4 where developed based on the composition of the original simulant sludges and a table of ingredients developed following the literature review in Section 2.4. In the following chapter the characterisation of these simulants is described and analysed.

4 Faecal Sludge Simulant Characterisation

This chapter provides details of the characterisation of the simulants developed for this thesis. Section 4.1 provides the methodology adopted for the characterisation tests. Details of the statistical analysis undertaken as part of the analysis of test results are provided in Section 4.2. The sources for the real FS characterisation data are discussed in Section 4.3. Comparison of the characterisation results for the FS simulants and the original simulants and real FS from a number of different containment types are discussed in Sections 4.4 and 4.5 respectively. Finally, Section 4.6 provides an overview of the analysis results and recommendation of the best performing simulant.

4.1 Methodology

In order to ensure repeatability and comparability of results from this work standard methods from soils and water and wastewater treatment were adopted. Alternative methods were recommended based on an extensive literature review of characterisation of real FS properties found in Appendix D. Availability of apparatus and time required where the main aspects taken into consideration for the recommendation of alternative methods. Appendix D should also prove useful as a database of reported tests for real FS by providing the reader with references for properties and methodologies. The primary aim was to produce a set of recommended test procedures which were simple and conducted with commonly available apparatus.

All FS simulants were characterised for the following properties determined as important in the literature review: moisture content; total solids; organics content; particle size distribution; ionic composition and concentration; degree of stabilisation; density; specific heat capacity; and thermal conductivity.

4.1.1 Moisture Content and Total Solids Concentration

Moisture content and total solids concentration were found according to the procedure outlined in *Standard Methods for the Examination of Water and Wastewater* (Greenberg et al., 1976) and BS 1377-2 (BSI, 1990). Furthermore, it is the same procedure as that used for every reference shown in Appendix D in which moisture content and/or total solids concentration is reported for real FS. This highlights the reliability and robustness of the method. Samples of known mass of each FS simulant were dried at 105°C for approximately 24 hours. Moisture content, w (%), was calculated as follows:

$$w = \frac{(A - B)}{(A - C)} \times 100\%, \tag{4.1}$$

where A (g) is the mass of wet sludge and container, B (g) is the mass of dry sludge and container and C (g) is the mass of the container. Total solids concentration, TS (%), was calculated from:

$$TS = \frac{(B - C)}{(A - C)} \times 100\%, \tag{4.2}$$

where all terms are as defined previously. Due to the ease of measurement for moisture content and total solids and the size of the oven available for drying, fifteen samples of each simulant were tested.

4.1.2 Organics Content

The organics content of the FS simulants was determined using volatile solids concentration. Volatile solids concentration was recommended as it is used as a measure of stabilisation since volatile solids are assumed to be composed of readily degradable organic matter (Strande et al., 2014). The ease and speed of the procedure, relatively low margin for user error and apparatus availability were further reasons for recommending volatile solids concentration. In accordance with the procedure outlined in (Greenberg et al., 1976) dried FS simulant samples were ignited in a furnace at 550°C for two hours. Volatile solids concentration, VS (g g dry sample^{-1}), was calculated from:

$$VS = \frac{(D - E)}{(D - F)}, \tag{4.3}$$

where D (g) is the mass of dry sludge and crucible before ignition, E (g) is the mass of dry sludge and crucible after ignition and F (g) is the mass of the crucible. The ratio of volatile solids to total solids was also reported to estimate degree of stabilisation, as it is indicative of the "relative amount of organic matter and the biochemical stability of FS" (Strande et al., 2014). Fixed solids, or ash, was calculated from:

$$Ash = \frac{(E - F)}{(D - F)} \times 100\%, \tag{4.4}$$

where all terms are as defined previously.

4.1.3 Particle Size Distribution

Particle size distribution was measured by laser diffraction, or low angle laser light scattering (LALLS), in keeping with the procedure outlined in BS ISO 13320 (BSI, 2010). A Malvern

Mastersizer 3000 particle size analyser with a size range of 0.01µm – 3500µm was used by PRG (2014). At the University of Bath analysis was conducted using a Malvern Mastersizer X with a range of 1.2µm – 600µm. The use of a different type of Mastersizer was not considered to be a significant issue as both machines operate on the same principles. Analysis of samples is done using the Fraunhofer model and Mie theory which "allows completely accurate results over a large size range (0.02 to 2000µm typically" (Rawle, n.d.). Simulant sample preparation for the tests performed at the University of Bath was prepared in accordance with the manufactures instructions. A slurry was prepared by mixing 50g of FS simulant with 100ml of distilled water. The slurry was stirred for 2 minutes to ensure even dispersion of particles. To prevent the slurry settling out it was continually stirred with a pipette which was simultaneously filled to add a sample to the small volume dispersion unit of the Mastersizer. Analysis was conducted in triplicate and in accordance with the manufacturer instructions. It was acknowledged that a Mastersizer, or equivalent apparatus, may not be readily available in some situations thus the dry sieving method recommended by Koottatep *et al* (2012) and outlined in BS EN ISO 17892-4 (BSI, 2016b) was considered as a possible alternative. Prior to analysis samples each FS simulant were dried at 105°C for approximately 24 hours. This was to ensure that the particles in the FS simulant samples were separated out from each other and having a dry sample would guarantee the actual weight of the simulant (Koottatep et al., 2012).

4.1.4 Ionic Composition and Concentration

To measure the ionic composition and concentration pH was recommended, as it can be used as an indicator of sludge stabilisation: pH levels for digested wastewater sludge tend to be more alkaline whilst primary sludge tends to have more acidic pH levels (Kopp and Dichtl 2001). The pH of each simulant was determined using a pH meter in accordance with the method recommended by PRG (Reddy, 2017). A Hanna HI 8424 pH meter was used to perform pH measurements at the University of Bath six times for each simulant. Each FS simulant sample was prepared by mixing 10g of simulant with 10g distilled water in a 50ml beaker and stirred continuously for 5 minutes.

4.1.5 Density

Dry density of samples was determined using an Accupyc™ 1330 pycnometer with three purges and three runs in accordance with manufacturer instructions and with BS EN ISO 17892-3 (2015). It was recommended to determine dry density as moisture content would is known to affect the bulk density. Furthermore, density results for the pycnometer benefit from a high degree of accuracy (Basu, 2010). Penn *et al* (2018) used the displacement

technique to determine the bulk density of the simulants (SE80, SB80, SE65, SB65) developed but recommended the use of a pycnometer to reduce uncertainty. Koottatep *et al* (2012) also recommend the use of a pycnometer for the determination of density. Other methods used for the determination of density can be found in Appendix D.

4.1.6 Thermal Properties

There was no APHA Standard Method for the determination of the thermal properties of wastewater sludge; therefore, it was necessary to pursue other methodologies. Only five sources were found which reported thermal conductivity or specific heat capacity values for real FS (see Appendix D). Of those five sources only one – Koottatep *et al* (2012) – used a different method to that used by PRG (Reddy, 2017). The thermal properties of simulant and real FS reported by Zuma *et al* (2015), Makununika (2016), Velkushanova *et al* (2017) and Septien *et al (2018)* were determined using a C-Therm TCi small volume test kit (Reddy, 2017). The C-Therm TCi employs the modified transient plane source technique for the determination of thermal conductivity in accordance with ASTM standards (C-Therm Technologies, 2020). The apparatus can measure thermal conductivity directly whilst specific heat capacity may be determined indirectly. Koottatep *et al* (2012) reported the use of a PA Hilton B480 Thermal Conductivity Analyser to measure thermal conductivity. The B480 was developed to determine the thermal conductivity of typical building materials and is based on the ISO 8301 (ISO, 1991) standard for the determination of steady state thermal conductivity. Neither of the aforementioned instruments was available at the University of Bath or via the GW4 Equipment Database. Thus, an alternative technique for the determination of thermal conductivity was sought.

Based on the speed and accuracy of measurement, the non-steady-state probe method was recommended to determine the thermal conductivity of the FS simulants in accordance with BS EN ISO 22007-1 (BSI, 2018) and ASTM D5334 (ASTM, 2014). A Hukesflux TP02 probe was used as it had previously been found suitable for analysing sludges (Hukesflux Thermal Sensors, 2003). 250ml of simulant sludge sample was allowed to come to thermal equilibrium and then stirred continuously for 3 minutes to ensure homogeneity and eliminate any bubbles which may have formed in the sample. Heating time was set to 180 seconds on low to avoid errors due to local moisture transport/evaporation by excessive heating (Hukesflux Thermal Sensors, 2003). Results were calculated using the tpsys02calculationsv1001.xlsx spreadsheet provided by Hukesflux. Six tests were conducted per simulant.

Specific heat capacity of the simulants was measured by differential scanning calorimetry in accordance with BS EN ISO 11357-1 (BSI, 2016a) and ASTM E1269-11 (ASTM, 2011). Scanning

differential calorimetry was also the method used by Koottatep *et al* (2012) to analyse real FS. Tests were conducted using a Setaram microSC microcalorimeter. Samples were heated from 10°C to 100°C with a heating rate of 1.2 K min^{-1} and a 10 minute step per 10°C. It was previously discussed in Chapter 4 that thermal properties change with temperature and moisture content. Therefore, tests were conducted with dried simulant samples and the results were then be extended to wet sludge using the method outlined by Vaxelaire and Puiggali (2002). The specific heat capacity of wet sludge, Cp_{sl}, was determined as follows:

$$Cp_{sl} = \frac{Cp_s + wCp_w}{1 + W}, \tag{4.6}$$

where Cp_s (J kg^{-1} K^{-1}) is the specific heat capacity of the dry solid, Cp_w (J kg^{-1} K^{-1}) is the specific heat capacity of water and w (g g^{-1}) is the moisture content of the simulant being tested. Each simulant was analysed four times.

4.1.7 Degree of Stabilisation

An estimation of the degree of stabilisation is provided using the ratio of volatile solids to total solids.

4.1.8 Summary of Characterisation Tests and Methodologies

A summary of the characterisation tests, methodologies followed, and apparatus required is shown in Table 4-1.

Table 4-1. Characterisation tests, methods and apparatus (adapted from Deering et al (2018))

	Test		Units	Methodology	Apparatus
	Moisture content	*w*	%	Greenberg *et al* (1976) APHA Standard Methods 208 G. Volatile and fixed matter in nonfiltrable residue and in solid and semisolid samples	Oven
Solids Concentration	Total solids	*TS*	%	Greenberg *et al* (1976) APHA Standard Methods 208 G. Volatile and fixed matter in nonfiltrable residue and in solid and semisolid samples	Oven
Organic Matter	Ash		%	Greenberg *et al* (1976) APHA Standard Methods 208 G. Volatile and fixed matter in nonfiltrable residue and in solid and semisolid samples	Furnace
	Volatile solids	*VS*	%	Greenberg *et al* (1976) APHA Standard Methods 208 G. Volatile and fixed matter in nonfiltrable residue and in solid and semisolid samples	Furnace
	Particle size distribution	*PSD*		BS EN ISO 13320 (2010)	Malvern X Mastersizer
				BS EN ISO 17892-4 (2016b)	Sieves
Ionic Composition and Concentration	pH			Greenberg *et al* (1976) APHA Standard Methods 424. pH value	Hanna H1 8424 pH meter
	Density	ρ	kg/m^3	BS EN ISO 17892-3 (2015)	AccuPyc 1330

	Test		Units	Methodology	Apparatus
Thermal Properties	Specific heat capacity	c	J/(kg·K)	ASTM E1269-11 (2011); BS EN ISO 11357-1 (2016a)	Setaram microSC microcalorimeter
	Thermal conductivity	λ	W/(m·K)	ASTM D5334 (2014); BS EN ISO 22007-1 (2018)	TP02 Hukseflux Thermal Sensor Temperature Sensor Type K; Campbell Scientific CR1000 datalogger

4.2 Real Faecal Sludge Data Sets

As well as comparison of the characterisation test results for the original PRG (2014) and Penn *et al* (2018) simulants, FS simulant results were compared with results for real FS from a variety of containment technologies. Two data sets from Velkushanova *et al* (2017) and Koottatep *et al* (2012) were used for these comparisons as these were the largest dataset available at the time of testing for results of real FS from a variety of containment technologies which encompassed type A and type B – stabilised and un-stabilised – sludge. This would allow comparison of simulant results with different types of sludge. The procedures followed to determine the properties of the FS simulants in this work are largely the same as those followed by Velkushanova *et al* (2017) and/or Koottatep *et al* (2012) thus ensuring comparability of results.

Velkushanova *et al* (2017) and Koottatep *et al* (2012) reported real FS results from South Africa and Thailand respectively. It has been noted previously that both climate and diet play significant roles in influencing real FS characteristics (Strande et al. 2014). As the original feasibility study for covered, USDBs was conducted in Bangladesh, and due to the lack of information of real FS from Bangladesh, it was necessary to rely on reported FS results from other locations. According the World Bank (2020) Bangladesh historically has a warm, humid climate and is one of the wettest countries in the world. This can significantly influence the characteristics of FS as noted by Bassan *et al* (2013). The mean annual precipitation is 2436.6mm and the mean annual temperature is 25.1°C (World Bank, 2020a). The general climate in South Africa is colder and drier with a mean annual temperature of 17.5°C and mean annual precipitation of 469.9mm in (World Bank, 2020b). The climate in Thailand is more comparable with Bangladesh as it is also tropical with a mean annual temperature of 26.3°C and mean annual precipitation of 1553.6mm (World Bank, 2020c).

According to the Food and Agriculture Organisation (FAO) of the United Nations (UN) (Bhattacharjee et al., 2007) the typical Bangladeshi diet is generally vegetarian and consists largely of rice. The Thai diet is very similar, also relying on rice as the most important food

source (FAO, 2014b). A more varied diet is likely in South Africa although a study by the Human Sciences Research Council (HSRC) (Shisana et al., 2013) found this was largely dependent on location (urban formal versus rural informal) and race. The results for real FS reported by Velkushanova *et al* (2017) were determined for samples of FS from various containment technologies in the eThekwini Municipality, Durban. It is likely that there will be variation in the characteristics of FS from both South Africa and Thailand in comparison to Bangladesh. However, it is not clear if the greatest influence in the variation will be caused by climate, diet or the containment technology used and user habits. The effect of diet in particular is difficult to quantify. Although it is commonly known to affect the characteristics of fresh excreta (Rose et al., 2015), its effect on more stabilised FS is yet to be researched.

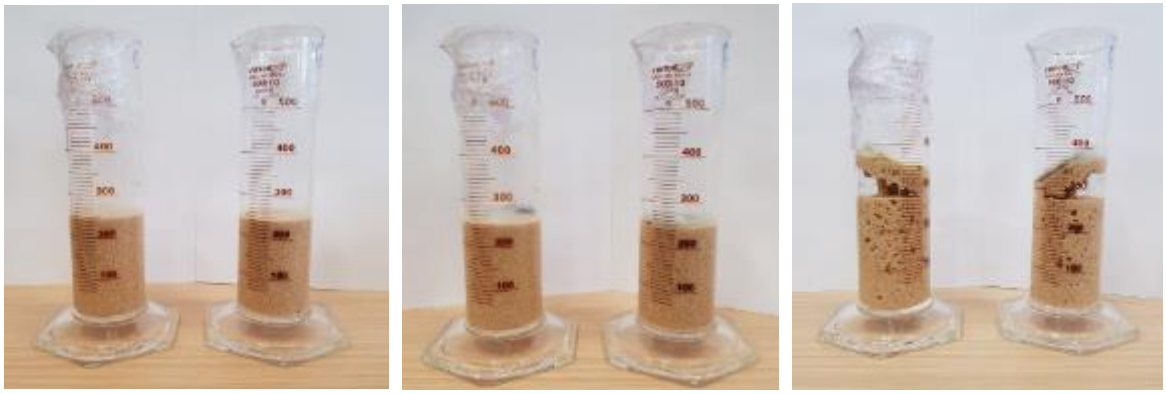
4.3 Statistical Analysis

Due to the small number of repeated tests for each characteristic, standard deviations and coefficient of variance could not be calculated as it was not possible to assume that data are normally distributed (Hibbert and Gooding, 2006; Schmid, 2014). Instead the median and interquartile range (IQR) were reported for the simulants (SS9, BM1-6, SE80 and ND1-4) tested at the University of Bath as these measures are more robust to outliers ad are considered suitable when it is not possible to use standard deviations. Outliers were identified using the interquartile range and k = 2.2 as recommended by Hoaglin and Iglewicz (1987). Outliers were not identified where six or fewer samples were tested as the margin of error was much greater due to the small sample size (Hoaglin and Iglewicz, 1987).

IQR and median values were calculated for the results presented by Koottatep *et al* (2012) as raw data was available; however, only averages were available for FS characteristics provided by Velkushanova *et al* (2017). Presenting the median and IQR for characterisation test results instead of, or as well as, average values is useful in helping to prevent severe under or over estimation of FS properties which will assist in the development of more representative simulants. Results of the characterisation tests are presented graphically in box plots and bi-directional charts. Note that the negative numbers on the bi-directional charts are indicative of direction on the chart (left hand side) only. Tables showing average, IQR and median values of all test results for simulants and real FS results (where applicable) may be found in Appendix E.

4.4 Comparison of Simulants Tested at Bath With Original PRG (2014) and Penn *et al* (2018) Simulants

The addition of brewers yeast to replace active yeast in simulants BM1 and BM2 to prevent bubble formation was found to be marginally successful. Samples of the simulants containing brewers yeast (BM3 and BM4) were observed over the course of eight days. It was noted that bubble formation was delayed from approximately 30 minutes to almost eight days as shown in Figure 4-1. The simulant samples expanded as bubbles formed and then collapsed as shown in Figure 4-1c. A thick, mouldy skin was seen to have formed on top of the simulant sludge samples. This was in keeping with observations of crust formation made during the thermal drying of wastewater and faecal sludges discussed in Chapter 2. Simulants BM5 and BM6 contained no yeast and no bubbling was observed over an eight-day period. This was taken as conclusive that bubble formation was caused by the presence of yeast. However, it was noted that both BM5 and BM6 appeared to rot and gave off a pungent smell within two to three days, thus rendering them unsuitable for drying experiments which would take at least ten days.



(a) *(b)* *(c)*

Figure 4-1. Photographs of BM3 showing delayed bubble formation: (a) Day 1, (b) Day 5, (c) Day 8 (Deering et al., 2018)

The characterisation test results for simulants SS9 and BM1-6 were compared with those reported for the original PRG (2014) simulant as shown in Table 4-2.

Table 4-2. Results of characterisation tests for faecal sludge simulants SS9 and BM1-6 tested at the University of Bath and those reported for the original PRG (2014) simulant where PRG is Pollution Research Group, SS is synthetic sludge, BM is Bath mix, w_{ave} (%) is average water content, TS_{ave} (%) is average total solids, VS_{ave} (g g dry sample^{-1}) is average volatile solids, λ_{ave} (W mK^{-1}) is average thermal conductivity and c_{ave} (J kg^{-1}K^{-1}) is average specific heat capacity at approximately 26.8°C

Simulant	**ρ_{ave} (kg m^{-3})**	**pH_{ave}**	**W_{ave} (%)**	**TS_{ave} (%)**	**VS_{ave} (g g dry sample^{-1})**	**Ash_{ave} (g g dry sample^{-1})**	**λ_{ave} (W mK^{-1})**	**c_{ave} (J kg^{-1}K^{-1})**
PRG[a]	1300.00	5.90	81.00	19.00	0.75	0.12	0.50	2700.51[b]
SS9	1344.06	5.21	80.49	19.51	0.82	0.18	0.05	2696.78
BM1	1342.70	5.26	81.02	18.98	0.83	0.17	0.05	2723.58
BM2	1351.19	5.26	81.00	19.00	0.83	0.17	0.06	2701.54
BM3	800.92	5.30	80.51	19.49	0.82	0.18	0.05	2609.35
BM4	1369.61	5.12	80.03	19.97	0.85	0.15	0.05	2637.04
BM5	1356.87	5.70	85.86	14.14	0.80	0.20	0.06	2792.67
BM6	1374.17	5.71	85.75	14.25	0.80	0.20	0.06	2856.27

[a] Results as reported in Velkushanova *et al* (2017)
[b] Specific heat capacity at approximately 20°C (Septien, pers. comm.)

From the results shown in Table 4-2, it was concluded that the original simulant developed by PRG (2014) is robust to alterations as the results for SS9 and BM1-6 are relatively similar to those for the original simulant with regards to density, pH, moisture content, total solids, volatile solids, ash content and specific heat capacity. As shown in Table 4-2, the average volatile solids and average ash contents of the simulants tested at the University of Bath were slightly higher than the average values reported for the PRG simulant. This was potentially due to differences in the type of cellulose used or slight differences in other organic components. It is likely that user error with regards to measurement of components would have some impact on the actual proportions in each simulant recipe which would affect test results.

Results for specific heat capacity are relatively similar for the simulants tested at the University of Bath and the original PRG (2014) simulant. The temperatures at which they were determined were different as shown in Table 4-2, so some difference in specific heat capacity is to be expected. There is a significant difference in the results for thermal conductivity reported by Velkushanova *et al* (2017) and the simulants tested at the University of Bath. A potential reason for the significantly lower average thermal conductivity reported for simulants SS9 and BM1-6 could be the presence of air bubbles during testing. Air is known to have a very low thermal conductivity (approximately 0.025 W m^{-1} K^{-1}) and could have influenced the measurement significantly.

Other discrepancies are the much lower average density reported for BM3 and the lower average total solids content – and higher average moisture content – reported for simulants BM5 and BM6. The lower total solids content was likely due to the omission of yeast of any kind from the simulant recipes for BM5 and BM6. The much lower average density reported for simulant BM3 was due to the large spread in density measurements recorded. The raw data for the density measurements of simulant BM3 is shown in Table 4-3.

Table 4-3. Raw data for density measurement of faecal sludge simulant BM3 where M is mass

Sample	1	2	3	4	5	6	7	8	9
M_{pan} (g)	1.890	1.901	1.901	1.901	15.621	15.621	1.901	1.902	1.902
$M_{pan + sample}$ (g)	4.455	4.113	3.984	3.933	16.778	16.546	4.305	3.682	3.680
M_{sample} (g)	2.565	2.213	2.083	2.032	1.157	0.924	2.404	1.780	1.778
ρ (g cm^{-3})	1.365	1.365	1.365	0.688	0.201	0.165	0.684	0.685	0.690
ρ (kg m^{-3})	**1365.4**	**1365.5**	**1364.7**	**687.5**	**200.8**	**165.4**	**683.8**	**685.0**	**690.2**

The minimum density measured was 165.4 kg m^{-3} whilst the highest was 1365.5 kg m^{-3} which was more comparable with the density of the other simulants tested. A contributory factor to the large difference in densities measured may have been because it was necessary to test samples across two different days. Samples 1-4 were tested at the same time; however, it was necessary to test the remaining samples on another day. The usual pan used for testing was not available initially for the second set of tests and a heavier one was necessary. Samples 5 and 6 were measured using this different pan and the lowest density measurements were recorded; therefore, the pan may have been the issue in this case. However, the pan was standard and compatible for use with the apparatus and a calibration was performed. A second calibration was performed with the original pan when it was returned. Furthermore, measurements with the normal pan (samples 7-9) were still lower than the first three tests but comparable with sample 4, which was measured at the same time as samples 1-3. Given that simulants SS9 and the remaining BM mixes had similar densities it was assumed that the low-density values BM3 may have been errors due to sample preparation. If the lower values were ignored, the average density for simulant BM3 was 1365.2 kg m^3 which is comparable with the other simulants tested at the University of Bath.

The characterisation test results for FS simulant SE80 was compared with those reported for the original Penn *et al* (2018) simulant as shown in Table 4-4.

Table 4-4. Results of characterisation tests for faecal sludge *simulant SE80 tested at the University of Bath and those reported for the original Penn et al (2018) simulant where SE is sludge containing only yeast extract, w_{ave} (%) is*

average *water content, TS*$_{ave}$ (%) is average total solids, VS$_{ave}$ (g g dry sample^{-1}) is average volatile solids, λ_{ave} (W mK^{-1}) is average thermal conductivity, c_{ave} (J kg^{-1}K^{-1}) is average specific heat capacity at approximately 26.8°C and NR is not reported

Simulant	ρ_{ave} (kg m^{-3})	pH$_{ave}$	W$_{ave}$ (%)	TS$_{ave}$ (%)	VS$_{ave}$ (g g dry sample^{-1})	Ash$_{ave}$ (g g dry sample^{-1})	λ_{ave} (W mK^{-1})	c$_{ave}$ (J kg^{-1} K^{-1})
SE80	1308.24	5.30	82.45	17.55	0.91	0.09	0.06	2688.73
Penn *et al* (2018)	1070.00	5.40	80.00	20.65	0.88	NR	NR	NR

The results reported for simulant SE80 tested at the University of Bath are quite similar to those reported for the original simulant with the exception of density. As the components and composition of the original recipe for SE80 given by Penn *et al* (2018) were not altered in any way a potential reason for the difference is methodology. As discussed in Section 6.2 Penn *et al* (2018) determined bulk density using the displacement method whereas dry density at the University of Bath was determined using a pycnometer. As Penn *et al* (2018) reported bulk density not dry density, results and methodology should not be compared. Although the original recipe for simulant SE80 was not altered, slight differences in the ingredients used could be a cause of differences in average values of the properties reported in Table 4-4. User error in weighing of components also cannot be discounted; slightly varying proportions of the recipe would be expected to have some impact on test results.

In order to assess the representativeness of the simulants tested at the University of Bath, results of the characterisation tests were compared with those for real FS. These comparisons are the focus of the following section.

4.5 Comparison of Simulants tested at Bath with Real Faecal Sludge

As discussed previously stabilised sludge is more suitable to drying on USDBs therefore it is desirable to match real type B FS results to the simulants. Type B sludge was identified according to the volatile solids to total solids ratio.

4.5.1 Moisture Content and Total Solids

Moisture content and total solids were considered together as they were measured using the same test. Moisture content and total solids concentration were determined by drying in an oven at 105°C according to standard methods as described in Section 6.2. Boxplots of the moisture content and total solids of the FS simulants tested at the University of Bath are shown in Figure 4-2 and Figure 4-3 respectively. In terms of both moisture content and total solids the least variability in results was noted for simulant SE80. This may have been due to the very homogenous nature of the raw materials used in the recipe which were

predominantly chemicals sources from Sigma Aldrich. Variety in the moisture content and total solids results for the other simulants tested may have been attributable to the natural variation in some of the raw ingredients used, for example, hemp fibre, psyllium husk and compost. However, the variability in results for SS9, BM1-6 and ND3-4 is much lower than for simulants ND1 and ND2 as shown in Figure 4-2 and Figure 4-3. This was likely due to more significant solids-liquids separation in the aforementioned simulants which led to difficulty in ensuring samples were representative and homogenous. Batches of simulants ND1 and ND2 were stirred continuously to ensure homogeneity and the suspension of solids prior to sampling for moisture content/total solids tests. However, there was still more significant variability in results when compared with the other simulants tested. Simulants BM5 and BM6 had higher average moisture contents and lower total solids contents than simulants SS9 and BM1-4 as yeast was omitted from both recipes entirely.

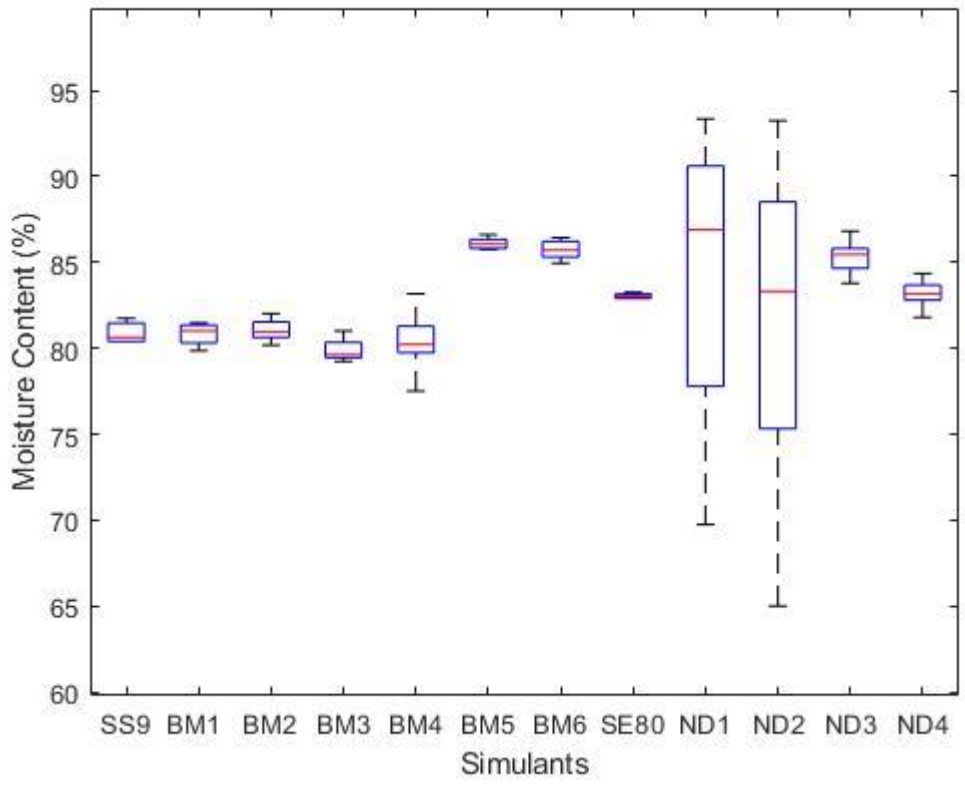


Figure 4-2. Boxplots for moisture content of faecal sludge simulants SS9, BM1-6 and ND1-4 *tested at the University of Bath*

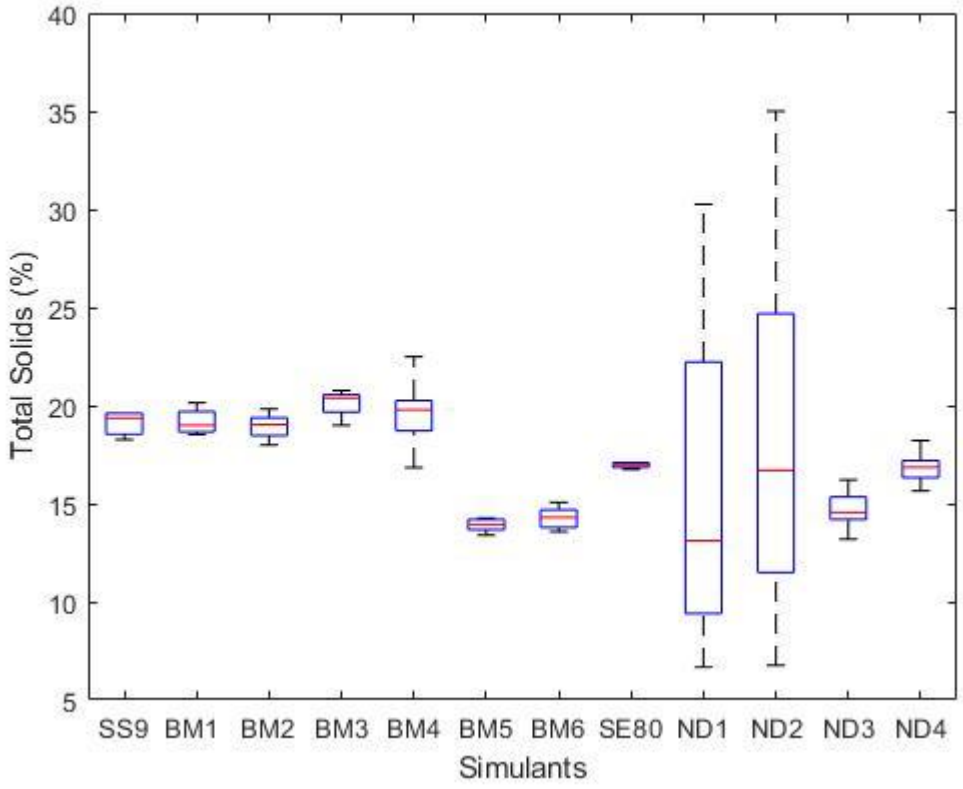
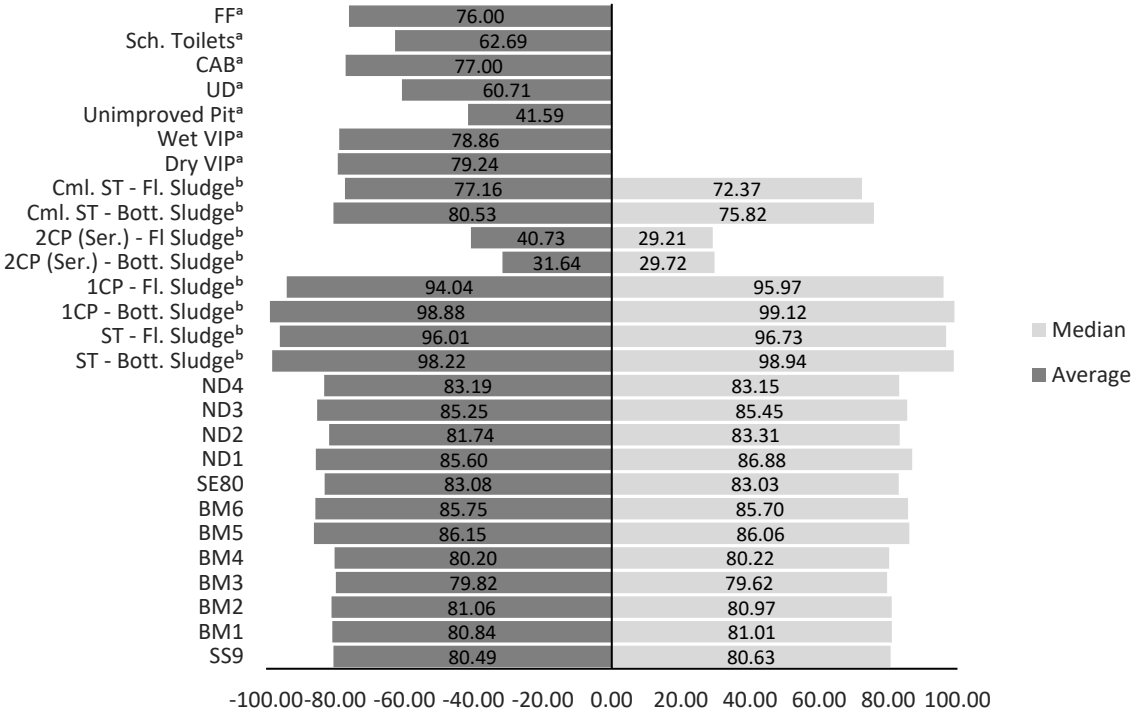


Figure 4-3. Boxplots for total solids of faecal sludge simulants SS9, BM1-6 and ND1-4 *tested at the University of Bath*

Average and median values for the moisture content of real FS and simulant FS are shown in Figure 4-4. The average moisture content of all the FS simulants tested at the University of Bath lies between 79.82% (BM3) and 86.15% (BM6). This is within the range of average values reported for the real FS shown in Figure 4-4. According to Koottatep *et al* (2012) blackwater and/or greywater was introduced by the users into the containment technologies they investigated i.e. septic tank, one cesspool, two cesspools in series and commercial septic tank. This would generally lead to less concentrated sludge as discussed previously and as indicated by the higher average values for moisture content in Figure 4-4 for containment technologies reported by Koottatep *et al* (2012). However, the average and median moisture contents reported for both fluidised and bottom sludge from two cesspools in series is noticeably lower than those reported for sludge from one cesspool, septic tanks and commercial septic tanks.

The moisture content of sludge from an unimproved pit latrine is also markedly lower than that of sludge from dry and wet ventilated improved (VIP) latrines, urine diversion (UD) toilets, community ablution blocks (CAB) or school toilets. The low average moisture contents may have been due to design of the containment technology or inputs into the system, but the precise cause is unknown as this information was not provided with the data. The average moisture content of all simulants SS9, BM1, BM2, BM4, BM5, BM6, SE80 and ND1-4 matched most closely with the average moisture content reported for bottom sludge

from commercial septic tanks. The average moisture content reported for simulant BM3 matched most closely with that reported for sludge originating from a dry ventilated improved pit with a percentage difference of 0.73%. All of the simulants tested at the University of Bath had average moisture contents which were similar to those reported for sludge originating from a dry or wet VIP latrine based on the percentage difference calculated as shown in Figure 4-4.



[a] Velkushanova *et al* (2017)
[b] Koottatep *et al* (2012)

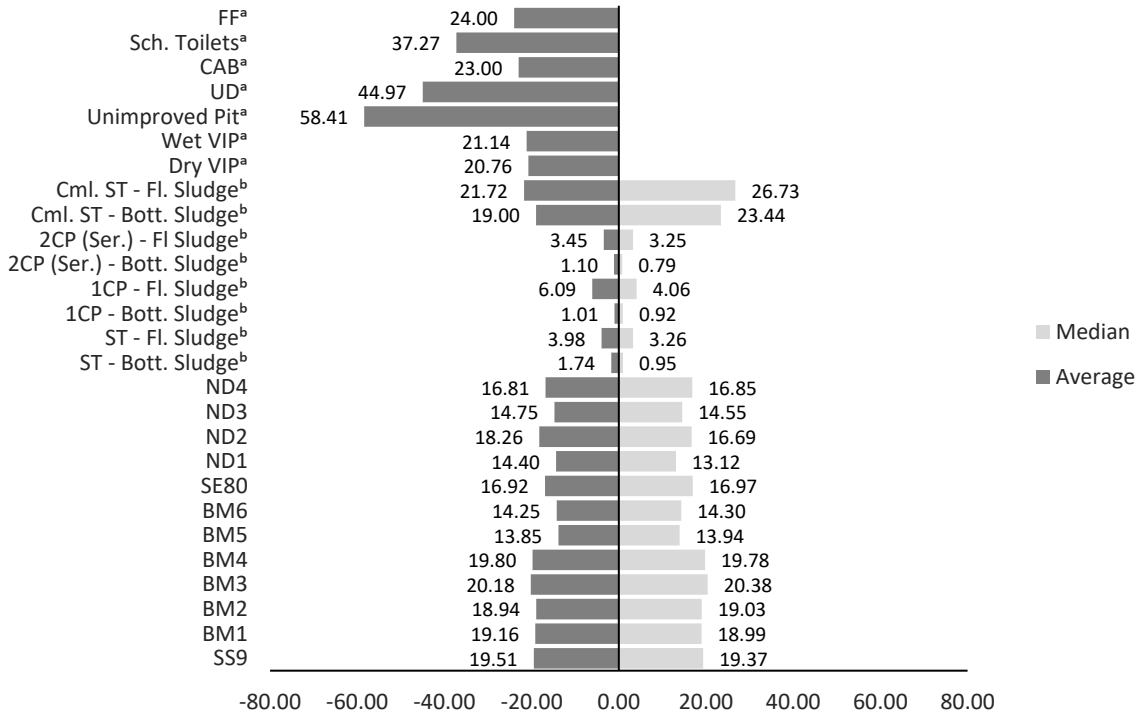
Figure 4-4. Bi-directional chart showing average (left hand side) and median (right hand side) moisture content of real faecal sludge and simulants SS9, BM1-6, SE80 and ND1-4 where Sch. Toilets is school toilets, CAB is community ablution block, UD is urine diversion, VIP is ventilated improved pit, Cml. ST is commercial septic tank, 2CP (Ser) is two cesspools in series, 1CP is one cesspool, ST is septic tank, Fl is fluidised and bott is bottom

Table 4-5. Percentage difference between average moisture content of simulants and most closely matching real faecal sludge where VIP is ventilated improved pit and Cml. ST – Bott. Sludge is bottom sludge from commercial septic tanks

	SS9	BM1	BM2	BM3	BM4	BM5	BM6	SE80	ND1	ND2	ND3	ND4
Dry VIP[a]	1.56	1.99	2.27	0.73	1.20	8.36	7.88	4.72	7.72	3.11	7.31	4.87
Wet VIP[a]	2.05	2.48	2.76	1.22	1.69	8.85	8.37	5.21	8.20	3.60	7.80	5.35
Cml. ST – Bott Sludge[b]	-0.05	0.38	0.65	-0.89	-0.41	6.75	6.27	3.11	6.10	1.49	5.70	3.25

[a] Velkushanova *et al* (2017
[b] Koottatep *et al* (2012

The values for total solids concentration reported by Koottatep *et al* (2012) were converted from mg L^{-1} into percentage by multiplying values by 0.0001 (Rees et al., 2015). As would be expected the due to the low moisture content, the total solids concentration of sludge originating from an unimproved pit was the highest as shown in Figure 6-6. The average values for FS originating from cesspools and non-commercial septic tanks were the lowest. This is reasonable considering the higher average moisture content reported for these sludges as shown in Figure 4-5. Simulants SS9, BM1, BM2, BM4, BM5, BM6, SE80 and ND1-4 all had average total solids concentrations which matched most closely with that reported for bottom sludge from a commercial septic tank. The simulant with the most similar average total solids concentration to bottom sludge from a commercial septic tank was simulant BM2 with a percentage difference of -0.30% as shown in Table 4-6 Simulants BM1 (0.88%), SS9 (2.66%) and ND2 (3.98%) were the next most representative. Simulant BM3 was most comparable to real FS from a dry VIP with a percentage difference of -2.83%. Therefore, in terms of moisture content and total solids concentration, the simulants tested at the University of Bath were found to be most representative of bottom sludge originating from a commercial septic tank or sludge originating from a dry VIP.



[a] Velkushanova *et al* (2017)
[b] Koottatep *et al* (2012)

Figure 4-5. Bi-directional chart showing average (left hand side) and median (right hand side) total solids of real faecal sludge and simulants SS9, BM1-6, SE80 and ND1-4 where Sch. Toilets is school toilets, CAB is community ablution block, UD is urine diversion and VIP is ventilated improved pit

Table 4-6. Percentage difference between average total solids concentration of simulants and most closely matching real faecal sludge where VIP is ventilated improved pit and Cml. ST – Bott. Sludge is bottom sludge from commercial septic tanks

	SS9	**BM1**	**BM2**	**BM3**	**BM4**	**BM5**	**BM6**	**SE80**	**ND1**	**ND2**	**ND3**	**ND4**
Dry VIP[a]	-6.20	-7.98	-9.16	-2.83	-4.72	-39.95	-37.16	-20.35	-36.18	-12.83	-33.86	-21.04
Wet VIP[a]	-8.04	-9.82	11.00	-4.67	-6.57	-41.72	-38.93	-22.17	-37.96	-14.66	-35.65	-22.86
Cml. ST – Bott Sludge[b]	2.66	0.88	-0.30	6.03	4.14	-31.37	-28.53	-11.54	-27.54	-3.98	-25.19	-12.24

[a] Velkushanova *et al* (2017
[b] Koottatep *et al* (2012

4.5.2 Organic Matter Content

Volatile solids content and ash content were determined by ignition in a furnace 550°C according to standard methods as described in Section 6.2. Boxplots of volatile solids content

and ash content are shown in Figure 4-6 and Figure 4-7. Very little variability was observed in the measured values of volatile solids for simulants SS9, BM2, BM4, BM5, BM6, SE80, ND3 and ND4 as shown in Figure 4-6. More variability was observed in the values calculated for simulants BM3 and ND1, but the most variable results were found for simulant ND2 as indicated by the large range and interquartile range shown in Figure 4-6.

Samples to determine volatile solids and ash of each simulant were prepared from dried samples following moisture content/total solids tests. It was necessary to crush the dried samples and place in crucibles suitable for the furnace. Although the crushed, dried simulants were mixed to ensure homogeneity, it is possible that some samples may have contained more or less organic matter than others when placed in the furnace. This is a potential reason for the variability of some of the results for the simulants tested at the University of Bath. Similar observations may be drawn with regards to ash content. Similar observations may made with respect to ash content: simulants SS9, BM2, BM4, BM5, BM6, ND3 and ND4 exhibited relatively little variability in terms of ash content as shown in Figure 4-7. Greater variability was observed for simulants BM1, BM3, ND1 and ND2. This is likely a direct consequence of the greater variability in volatile solids content. Due to the variability in results, the median values of volatile solids and ash content are expected to be more representative of the actual volatile solids and ash content of the simulants as the median is less susceptible to outliers.

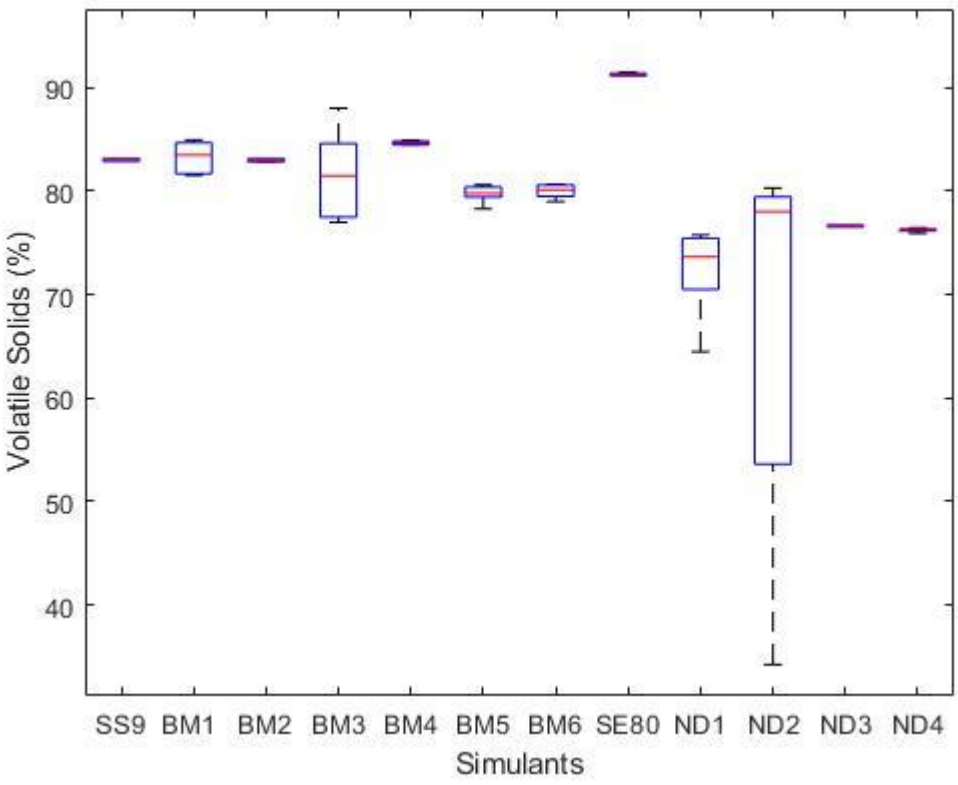


Figure 4-6.. Boxplots for volatile solids of faecal sludge simulants SS9, BM1-6 and ND1-4 *tested at the University of Bath*

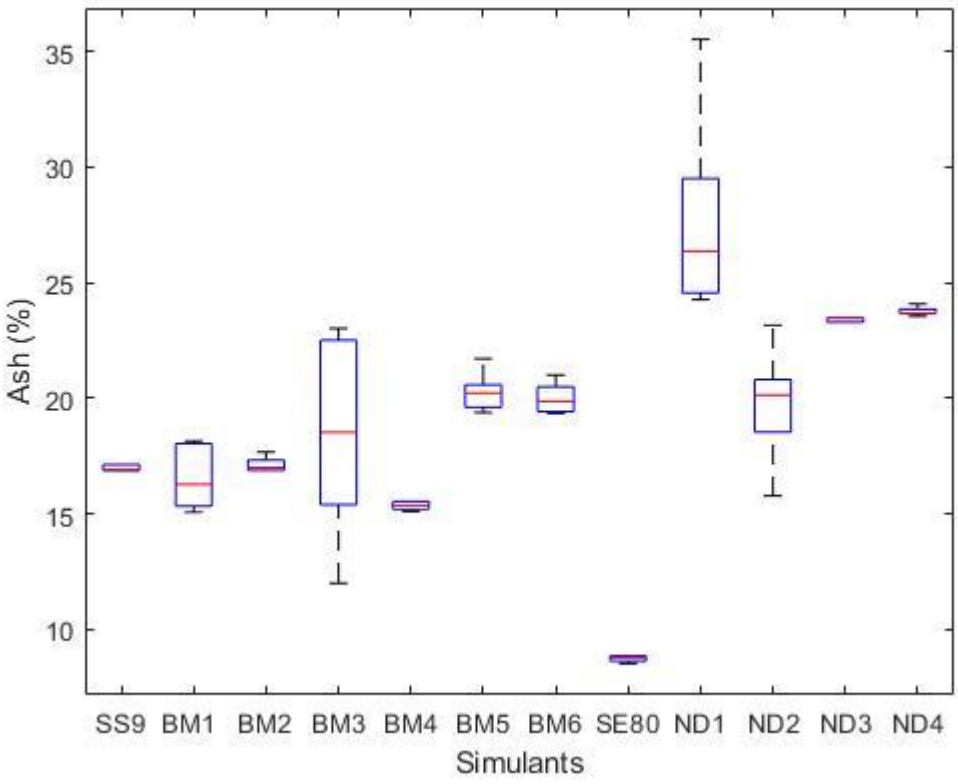
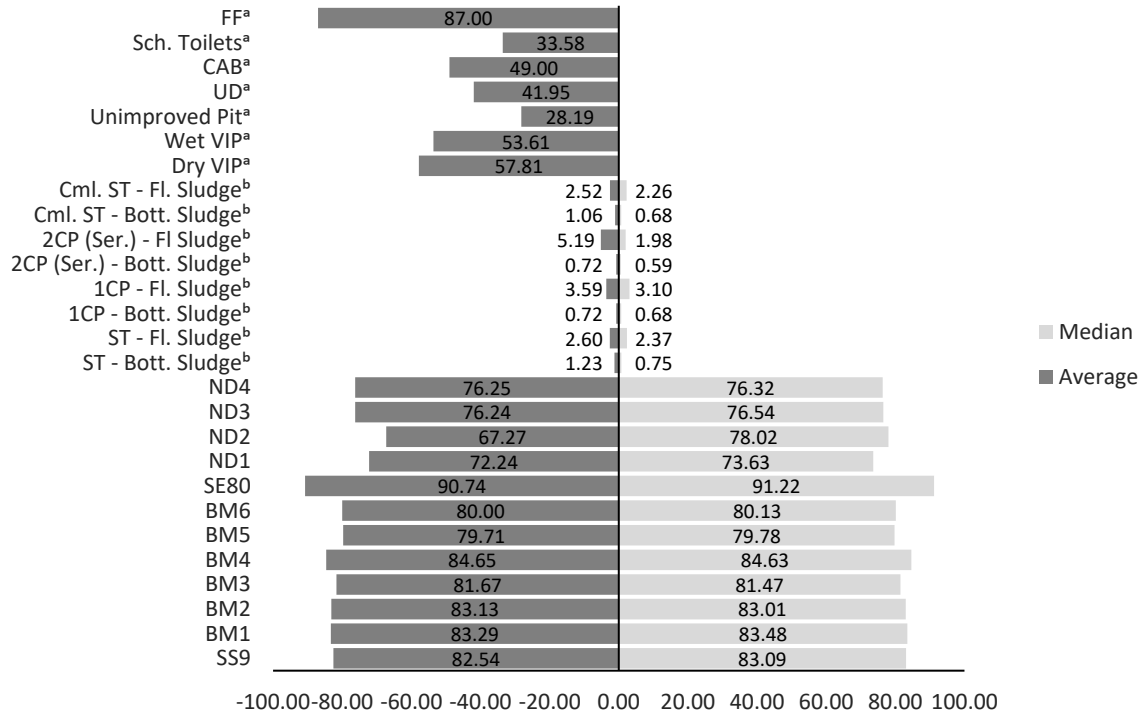


Figure 4-7. Boxplots for ash of faecal sludge simulants SS9, BM1-6 and ND1-4 *tested at the University of Bath*

As shown in Figure 4-8, the simulant with the highest median volatile solids content is simulant SE80 (91.22%). It is likely that the higher proportion of organics in the recipe for simulant SE80 (see Table 3-4) in comparison to the other simulant recipes, is the likely cause. Simulants SS9 and BM1-6 had similar median volatile solids contents. Given the similarity of the recipes used this was to be expected. In comparison with the other simulants tested, a greater proportion of simulants ND1-4 was made up of inorganics (Table 3-8) and therefore it was expected that the volatile solids contents were lower as shown in Figure 4-8.

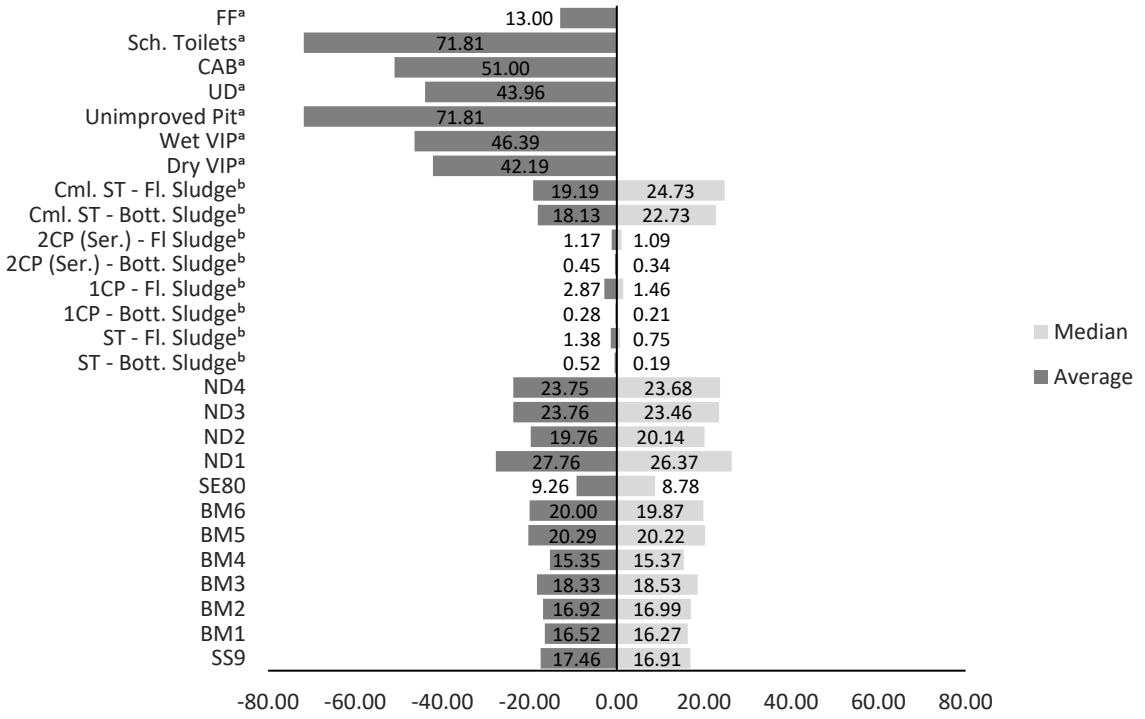
From Figure 4-8., it is clear that the average volatile solids content for all of the simulants tested is much higher than the average reported for real FS and are more comparable with fresh faeces. As noted by Ward *et al* (2017) this is likely because the FS simulants have not undergone digestion whilst real FS will undergo some stabilisation during storage in containment. According to Kopp and Dichtl (2001b) volatile solids content decreases during digestion of sludge; therefore, it is likely that the real sludges which have undergone the most digestion (stabilisation) are those reported by Koottatep *et al* (2012). The median values of all of the simulants tested matched most closely with the average reported for real FS from a dry VIP latrine since this sludge the highest volatile solids content of all the real FS. The simulant which was most representative of real FS originating from a dry VIP latrine was ND1, with a percentage difference of 24.08%. The percentage difference for all other simulants tested lay between 27.00% and 45.00%. Therefore, the simulants tested at the University of Bath are not representative of stabilised real FS in terms of volatile solids content.



[a] Velkushanova *et al* (2017)
[b] Koottatep *et al* (2012)

Figure 4-8. Bi-directional chart showing average (left hand side) and median (right hand side) volatile solids of real faecal sludge and simulants SS9, BM1-6, SE80 and ND1-4 where FF is fresh faeces, Sch. Toilets is school toilets, CAB is community ablution block, UD is urine diversion and VIP is ventilated improved pit

The ash content is a measure of the inorganics content of a sludge. Average and median values of ash content are shown in Figure 4-9. The average ash content of the simulants tested at the University of Bath was higher than that reported for fresh faeces with the exception of simulant SE80. Comparing the median values of the simulants with the median values calculated from real FS reported by Koottatep *et al* (2012), simulants ND3 and ND4 are most representative of bottom sludge originating from a commercial septic tank. Simulants ND3 and ND4 also match more closely than the other simulants tested with fluidised sludge from a commercial septic tank. Furthermore, simulant ND1 is a relatively close match with fluidised sludge from a commercial septic tank.



[a] Velkushanova *et al* (2017)
[b] Koottatep *et al* (2012)

Figure 4-9. Bi-directional chart showing average (left hand side) and median (right hand side) ash of real faecal sludge and simulants SS9, BM1-6, SE80 and ND1-4 where FF is fresh faeces, Sch. Toilets is school toilets, CAB is community ablution block, UD is urine diversion, VIP is ventilated improved pit, Cml. ST is commercial septic tank, 2CP (Ser) is two cesspools in series, 1CP is one cesspool, ST is septic tank, Fl is fluidised and bott is bottom

4.5.3 Particle size distribution

Particle size distribution of simulants was intended to be estimated by dry sieving and with a Malvern Mastersizer X as described in Section 6-2. To determine particle size distribution by dry sieving, sludge samples were dried in an oven at 105°C for 24 hours. It was expected that the simulants would crack and crumble in a similar manner to real FS when it is dried; however, this was not the case. Instead of cracking, all of the simulants tested at the University of Bath dried smoothly as shown in Figure 6-19. In order to perform sieving analysis, the dried simulant samples required crushing in a pestle and mortar. It was assumed this would not give a good representation of the particle size distribution of the sludge given that it would be similar to determining the size distribution of the ingredients in a cake batter following baking. Therefore, particle size distribution was not estimated by dry sieving.

The results of the particle size distribution of the simulants tested at the University of Bath with the Mastersizer X are shown in Table 6-18. The residual is of key importance as a residual less than one is indicative of a good fit of presentation data to measurement data. The lowest

residuals were found for simulants SS9 (0.348%) and BM3 (0.495%) whilst the highest were recorded for ND1 (1.562%) and ND4 (1.763%) as shown in Table 4-7. The simulants for which the residual was over one were excluded from any comparison with the particle size distribution of real FS.

Table 4-7. Mastersizer results of particle size distribution for samples with lowest residual of simulants SS9, BM1-6, SE80, ND1-4 where D[4,3] (µm) is the volume mean diameter, D(v,0.1) (µm) is the diameter below which 10% of particles are smaller, D(v,0.5) (µm) is the median particle diameter and D(v,0.9) (µm) is the diameter below which 90% of particles are smaller

Simulant	Residual (%)	D[4,3] (µm)	D(v, 0.1) (µm)	D(v,0.5) (µm)	D(v,0.9) (µm)	Uniformity
SS9	0.348	135.65	3.85	25.56	9.49	5.08
BM1	0.658	36.45	3.85	5.05	127.84	6.38
BM2	0.803	79.26	3.83	5.36	326.12	13.98
BM3	0.495	49.73	3.64	8.35	150.85	5.47
BM4	0.763	122.05	3.68	46.17	359.25	2.43
BM5	1.230	249.70	35.57	214.79	524.79	0.72
BM6	1.390	198.92	20.00	154.60	435.93	0.85
SE80	1.359	240.66	50.49	206.83	498.87	0.67
ND1	1.562	208.73	24.07	159.97	469.61	0.87
ND2	1.209	209.94	27.81	150.81	485.00	0.95
ND3	1.243	248.37	38.66	218.88	520.36	0.71
ND4	1.763	274.55	58.58	254.89	535.17	0.60

Vincent *et al* (2011, 2012) determined particle size distribution of septage by laser diffraction and reported average D(v,0.1) and D(v,0.9) values for septage of 3.30µm and 71.00µm respectively. The D(v,0.1) values of simulants SS9 and BM1-4 are comparable with the D(v,0.1) value reported by Vincent *et al* (2011, 2012). However, the D(v,0.9) values found for simulants BM1-4 were much higher than that reported by Vincent *et al* (2011, 2012) as shown in Table 4-7. Results could not be compared with real FS as the particle size distribution was not reported by for real FS by Velkushanova *et al* (2017). Results could also not be compared with particle size distribution data reported by Koottatep *et al* (2012) for real FS due to the different methodology used. This is because different techniques used to determine particle size distribution may measure different properties of the particles and therefore it is only recommended to compare measurements made using the same technique (Rawle, n.d.). The range of the 300mm lens (1.2µm - 600µm) appeared to have been adequate to estimate the particle size distribution of the simulant sludges as 100% of particles for all samples of each simulant fell below 600µm. However, it was not possible to compare results using the 1000mm lens with a range of 4.0µm-2000µm (used by PRG (Reddy 2017)) to assess the suitability because it was not available at the University of Bath.

The solids classification system used by Karr and Keinath (1978) has been discussed previously in Chapter 3 and is shown in Table 4-8. Karr and Keinath (1978) concluded that supra-colloidal particles (1 μm to 100 μm) had the greatest influence on the rate of sludge dewatering. The volume mean diameter of simulants SS9 and BM4 was found to be greater than 100μm whilst the volume mean diameter for simulants BM1-3 lay between the 1μm-100μm range for supra-colloidal particles given by Karr and Keinath (1978). This would imply that the simulants are unlikely to dewater well. As the residuals recorded indicate a poor fit for simulants BM5, BM6, SE80 and ND1-4 no conclusions could be drawn with regards to the solids fractions in these simulants.

Table 4-8. Solids classification system (Karr and Keinath, 1978)

Solids Fraction	**Size (μm)**
Settleable	≥100
Supra-colloidal	1-100
True colloidal	0.001-1
Dissolved	≤0.001

4.5.4 Ionic Composition and Concentration

A Hanna HI 8424 pH meter was used to determine pH values for all of the simulants tested at the University of Bath. There was very little variability in the pH results for each simulant as in Figure 6-6. The pH values for simulants SS9 and BM1-4 were relatively similar between 5.1 and 5.3. This was to be expected as the recipes used were very similar. The omission of yeast entirely in simulants BM5 and BM6 may have been the cause of the higher pH in comparison with simulants SS9 and BM1-4. There was virtually no variability exhibited in the pH values recorded for simulants SE80 and ND2 as shown in Figure 6-6 and the results were comparable with those for simulants SS9 and BM1-4. Simulants ND3 and ND4 exhibited the highest pH. The lower pH of ND1 in comparison to ND2-4 may have been due to the omission of ammonium carbonate from the recipe (see Table 3-8). None of the simulants tested had a pH above neutral (7) and the most acidic simulant was BM4.

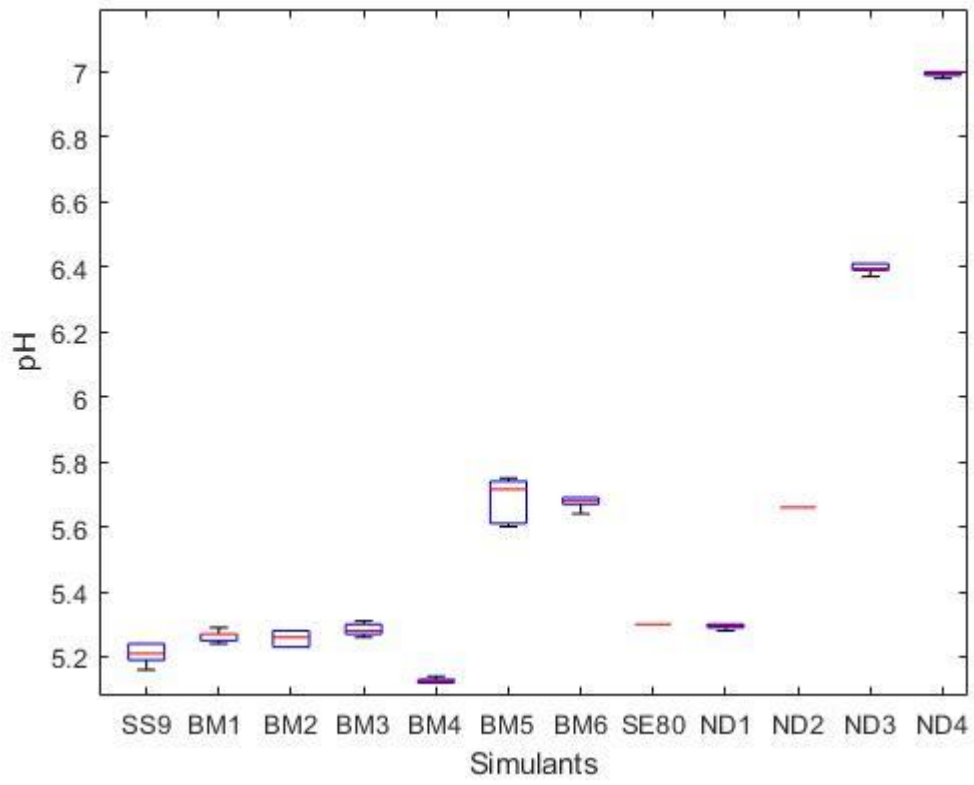
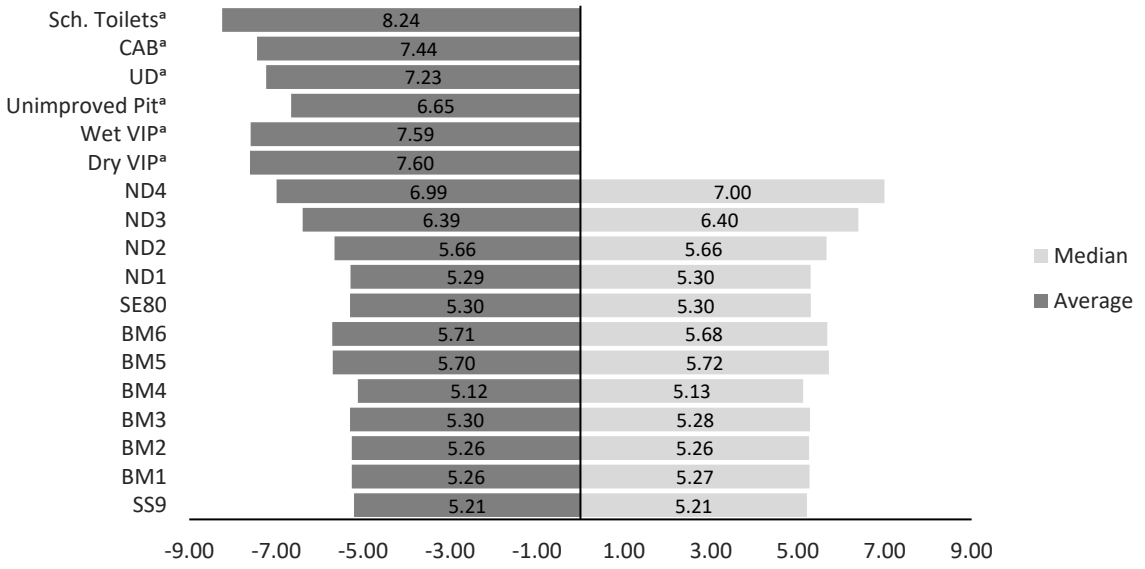


Figure 4-10. Boxplots for pH of faecal sludge simulants SS9, BM1-6 and ND1-4 *tested at the University of Bath*

As shown in Figure 4-10, average pH values for real FS was between 6.65 (unimproved pit) and 8.24 (school toilets) (Velkushanova et al., 2017). Higher pH has been found to be associated with less stabilised sludge (Ward et al., 2019). Based on this finding FS originating from unimproved pits is potentially more stabilised than real FS from the other sources/containment technologies given in Figure 4-10. Simulant ND3 had the closest average pH to real FS from an unimproved pit (largely stabilised sludge) whilst simulant ND4 was most similar to the average pH reported for real FS from a UD toilet (partially stabilised). The average pH for simulants SS9, BM1-6 and ND1-2 was much lower than for real FS. It was not possible to compare the median values of pH for the simulants with real FS as the raw data was not available and median values were not reported. The median values reported for the FS simulants were very similar to the average values reported as shown in Figure 4-11.

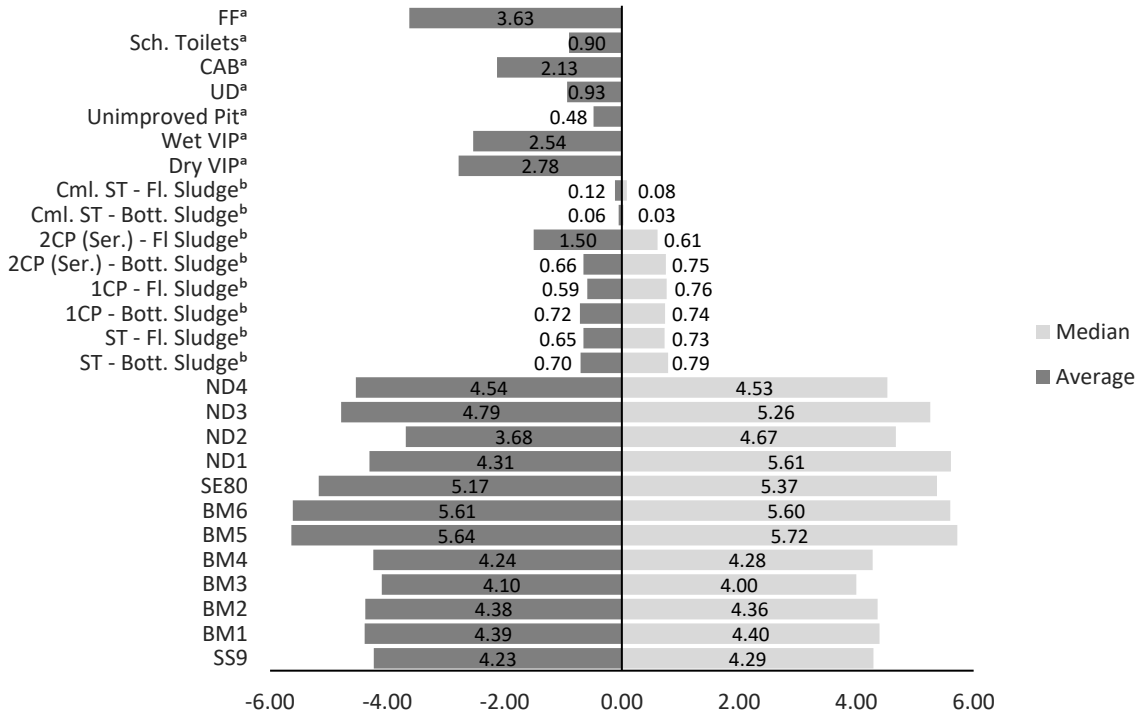


[a] Velkushanova *et al* (2017)

Figure 4-11. Bi-directional chart showing average (left hand side) and median (right hand side) pH of real faecal sludge and simulants SS9, BM1-6, SE80 and ND1-4 where Sch. Toilets is school toilets, CAB is community ablution block, UD is urine diversion and VIP is ventilated improved pit

4.5.5 Degree of Stabilisation

The average volatile solids to average total solids and median volatile solids to median total solids ratios were calculated for each simulant tested at the University of Bath as well as for real FS reported by Velkushanova *et al* (2017) and Koottatep *et al* (2012). According to Penn *et al* (2018) the volatile solids to total solids ratio for samples of real FS collected during discharge at treatment facilities generally ranges between 0.43 and 0.73 for more stabilised sludge. As would be expected, given that the simulants are not stabilised, the volatile solids to total solids ratios for all of the simulants tested at the University of Bath was much higher than those calculated for real FS as shown in Figure 4-12. The lowest ratio was found for simulant BM3 and the highest ratio was that calculated for simulant BM5. Although the volatile solids to total solids ratios for the simulants are not comparable to those for real FS, Figure 6-11 is useful to estimate the expected stability of the real sludge reported by Velkushanova *et al* (2017) and Koottatep *et al* (2012).



[a] Velkushanova *et al* (2017)
[b] Koottatep *et al* (2012)

Figure 4-12. Ratio of volatile solids to total solids for real faecal sludge and simulants SS9, BM1-6, SE80 and ND1-4 where FF is fresh faeces, Sch. Toilets is school toilets, CAB is community ablution block, UD is urine diversion, VIP is ventilated improved pit, Cml. ST is commercial septic tank, 2CP (Ser) is two cesspools in series, 1CP is one cesspool, ST is septic tank, Fl is fluidised and bott is bottom

A rough estimation of the stability of the real FS results reported by Velkushanova *et al* (2017) and Koottatep *et al* (2012) based on the volatile solids to total solids ratio is shown in Table 4-9. The ratios reported according to the median total solids and volatile solids were used to determine stability for real FS reported by Koottatep *et al* (2012) to reduce over or under estimation due to outliers. Fresh faeces were assumed to be raw and the sludge with the lowest volatile solids to total solids ratio was assumed to be stabilised. Percentage difference was used to estimate stabilisation of FS with respect to fresh faeces as shown in Table 6-17. Similar ratios were grouped together into raw (fresh faeces), partially raw, partially stabilised, largely stabilised and stabilised. As shown in Table 4-9 partially raw faecal sludge originated from dry and wet ventilated improved pit latrines and community ablution blocks. Partially stabilised FS originated from urine diversion toilets, school toilets, septic tanks and cesspools. Largely stabilised sludge included fluidised sludge from septic tanks and two cesspools in series as well as sludge from unimproved pit latrines. Fluidised and bottom sludge from commercial septic tanks was the most stabilised sludge according to the volatile solids to total solids ratios.

Table 4-9. Estimated stabilisation of real faecal sludge from a variety of containment technologies and sources where FF is fresh faeces, Sch. Toilets is school toilets, CAB is community ablution block, UD is urine diversion, VIP is ventilated improved pit, Cml. ST is commercial septic tank, 2CP (Ser) is two cesspools in series, 1CP is one cesspool, ST is septic tank, Fl is fluidised, bott is bottom, VS is volatile solids, TS is total solids and % Diff. is percentage difference

Containment technology/ Source	VS:TS	% Diff.	Degree of Stabilisation
FF[a]	3.63	0.00	Raw
Dry VIP[a]	2.78	-26.22	Partially raw
Wet VIP[a]	2.54	-35.38	
CAB[a]	2.13	-51.94	
UD[a]	0.93	-118.12	Partially stabilised
Sch. Toilets[a]	0.90	-120.36	
ST - Bott. Sludge[b]	0.79	-128.52	
1CP - Fl. Sludge[b]	0.76	-130.42	
2CP (Ser.) - Bott. Sludge[b]	0.75	-131.36	
1CP - Bott. Sludge[b]	0.74	-132.44	
ST - Fl. Sludge[b]	0.73	-133.12	Largely stabilised
2CP (Ser.) - Fl Sludge[b]	0.61	-142.49	
Unimproved Pit[a]	0.48	-153.00	
Cml. ST - Fl. Sludge[b]	0.08	-190.87	Stabilised
Cml. ST - Bott. Sludge[b]	0.03	-196.80	

[a] Velkushanova *et al* (2017)
[b] Koottatep *et al* (2012)

4.5.6 Density

Density of the FS simulants tested at the University of Bath was measured using an Accupyc™ 1330 pycnometer. Box plots of FS simulants SS9, BM1, BM2, BM4-6, SE80 and ND1-4 are shown in Figure 6-4. Simulant BM3 was excluded from Figure 4-13 due to the very large spread in values which rendered the box plots for all other simulants difficult to read. Possible reasons for the large spread in density values for simulant BM3 have been discussed previously. Simulants SS9, BM1, BM2, BM4, BM5 and BM6 were based on the same original recipe developed by PRG (2014) and exhibited similar densities with little variation as shown in Figure 6-4. Of the simulants based on the recipe developed by PRG (2014) the highest

density value was recorded for simulant BM6. Simulant BM4 had the smallest variation in measured density values. The greatest range and interquartile range were found for simulant ND3 as shown in Figure 6-4. Simulants ND1 and ND2 also had relatively large interquartile ranges in comparison with the other simulants tested.

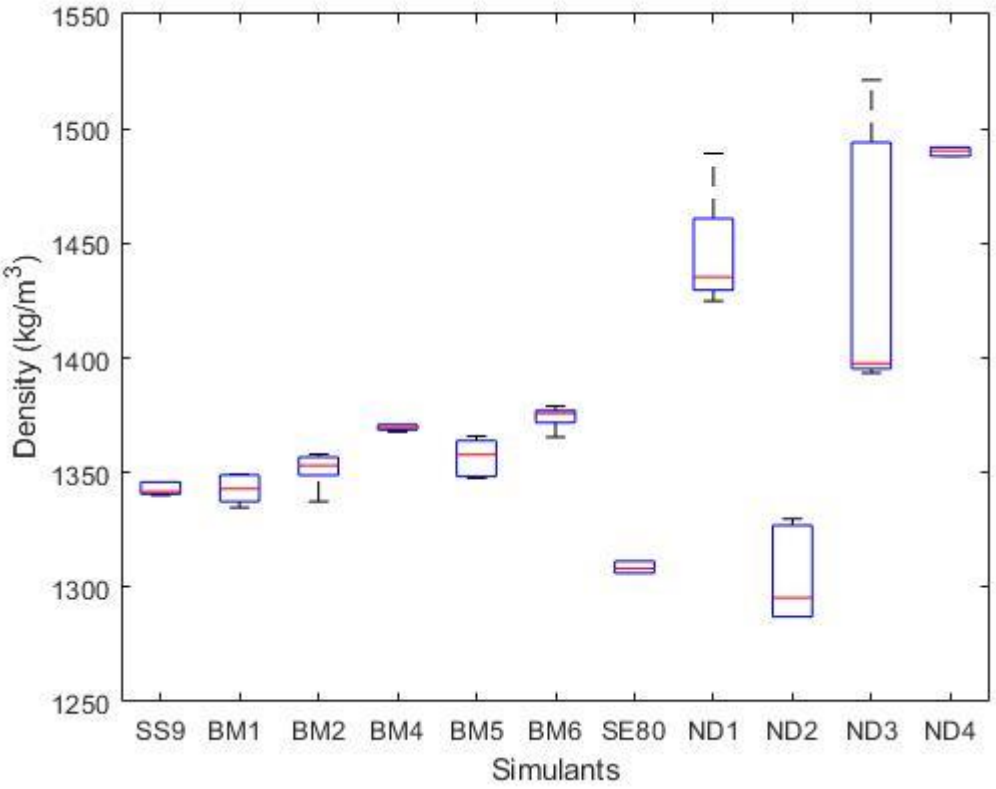
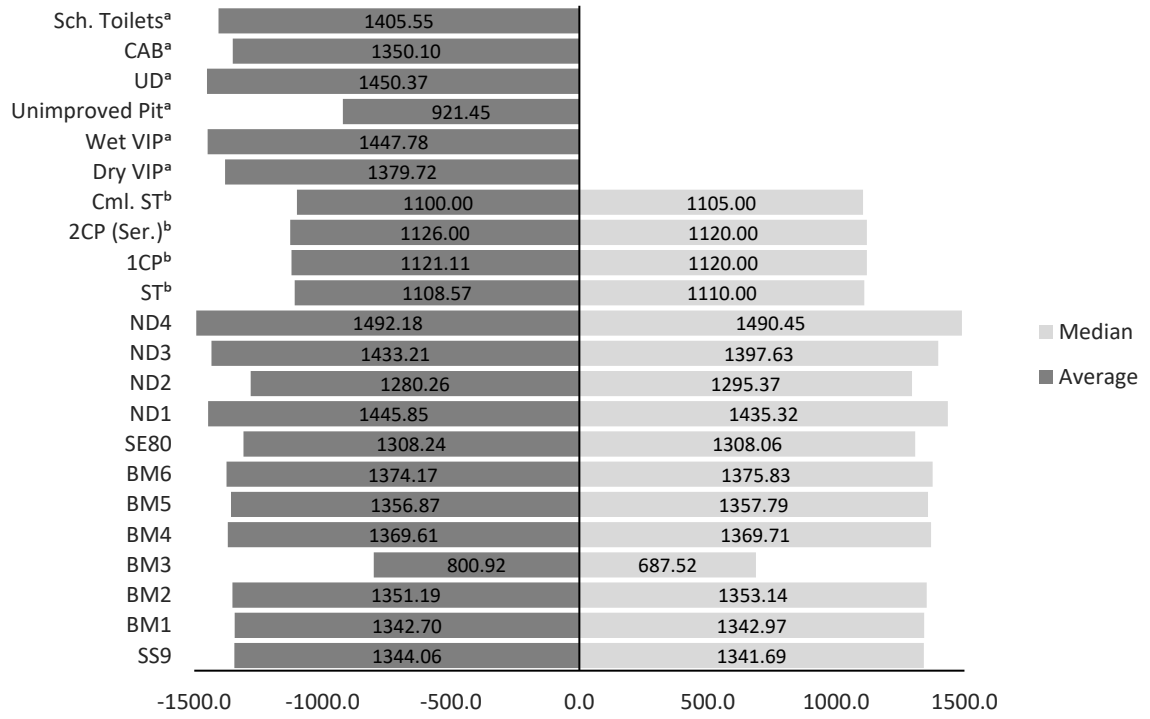


Figure 4-13. Boxplots for density of faecal sludge simulants SS9, BM1, BM2, BM4, BM5, BM6 and ND1-4 tested at the University of Bath

As shown in Figure 4-14 here was large variation in the average density of sludge from different sources reported by Velkushanova *et al* (2017). The lowest average density was recorded for FS from an unimproved pit latrine (921.45 kg m^{-3}) and the highest was 1447.78 kg m^{-3} for FS originating from a wet VIP latrine. The average values reported by Koottatep *et al* (2012) were similar to each other and ranged between 1126.00 kg m^{-3} (two cesspools in series) and 1100.00 kg m^{-3} (commercial septic tank). The lower variation is likely due to the similarity in containment technologies, as all were types of septic tank.



[a] Velkushanova *et al* (2017)
[b] Koottatep *et al* (2012)

Figure 4-14. Bi-directional chart showing average (left hand side) and median (right hand side) density of real faecal sludge and simulants SS9, BM1-6, SE80 and ND1-4 where Sch. Toilets is school toilets, CAB is community ablution block, UD is urine diversion, VIP is ventilated improved pit, Cml. ST is commercial septic tank, 2CP (Ser) is two cesspools[12] in series, 1CP is one cesspool and ST is septic tank

Apart from the average density reported for BM3, all average density values reported for the simulants tested at the University of Bath lie within the range reported for real sludge of between 921.45 kg m^{-3} for FS from an unimproved pit and 1450.37 kg m^{-3} for FS from a urine diversion latrine. Simulant ND2 had the closest average density to the results for real FS reported by Koottatep *et al* (2012) i.e. partially stabilised to stabilised sludge. Excluding BM3 from the comparison, simulant ND2 also had the closest match in average density for real FS from an unimproved pit (largely stabilised sludge). The average density of simulant BM6 matched closely with that reported for real FS from a dry VIP whilst the average density for simulant ND1 matched closely with that reported for real FS from a wet VIP. The sludge originating from both containment technologies was determined to be partially raw in Section 4.5.5 Average density of simulant ND1 was also the closest to the density of FS from a UD toilet (partially stabilised). Simulant SS9 had a similar density to that reported for real FS from CABs (partially raw) and FS from school toilets (partially stabilised) had an average density relatively similar to that of simulant ND3. The median values recorded for the

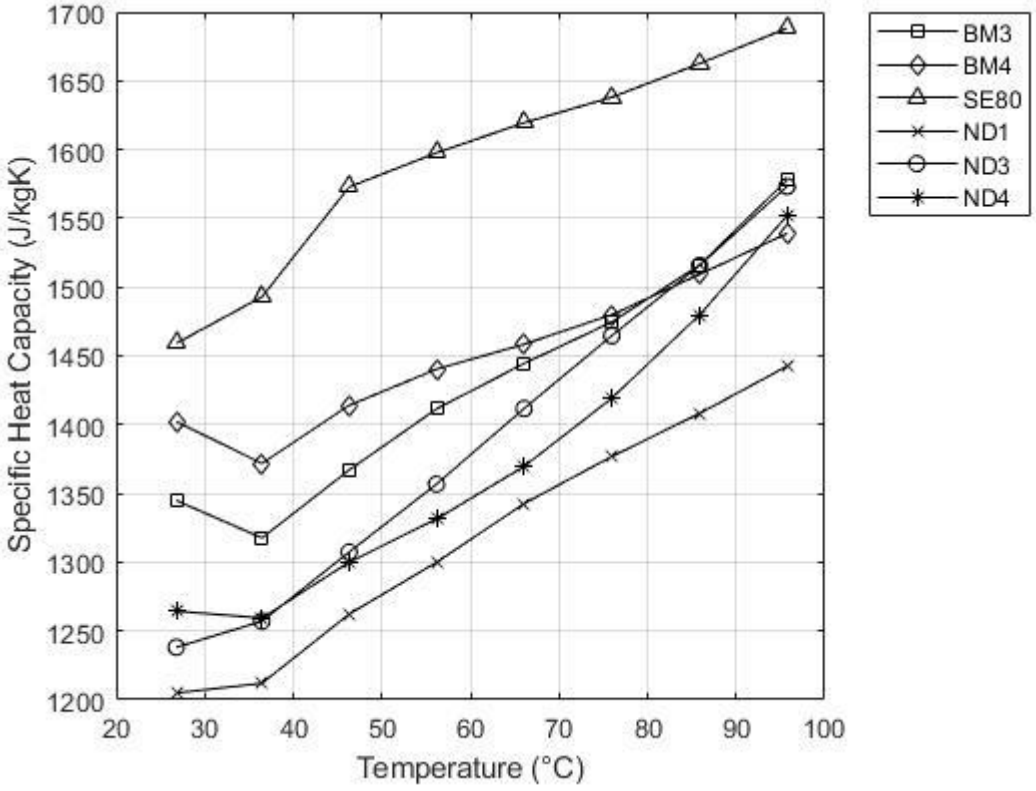
[12] According to Strande *et al* (2018) a cesspool is a partially lined septic tank.

simulant sludges were all higher than those calculated for the results reported by Koottatep *et al* (2012).

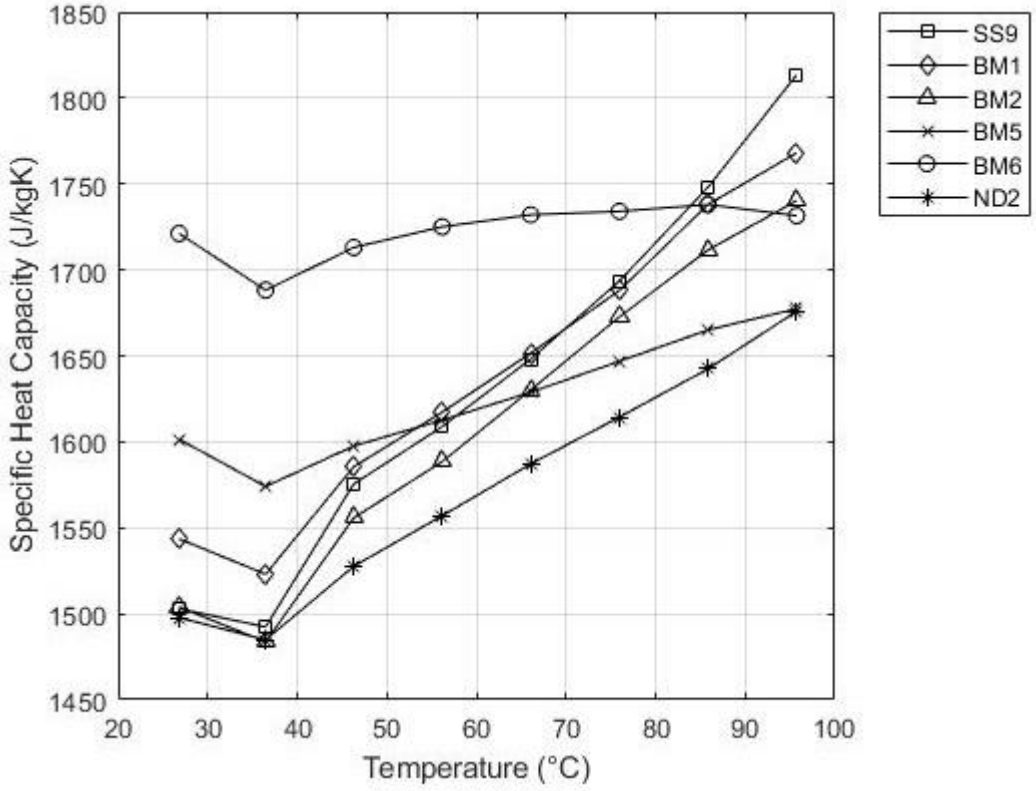
Based on density values, Radford *et al* (2015) made two sludge clusters. Sludge originating from unimproved pit latrines and septic tanks formed one cluster and sludge originating from VIP latrines, UD TOILETs, school toilets and CABs formed the other. These groupings appear to be applicable to the real FS average density results shown in Figure 6-5. The cluster containing sludge from unimproved pit latrines and septic tanks could be extended to include sludge from cesspools. Assuming that the unimproved pit and septic tank/cesspool grouping is indicative of the average density of stabilised sludge, then the simulants tested at the University of Bath do not fall into this group and may not be representative of the density of more stabilised sludge. The simulant with the lowest density, and therefore most comparable to that of stabilised sludge was ND2.

4.5.7 Thermal properties

Specific heat capacity of each simulant was tested as described in Section 4.1.6 . Some variability in the results of each simulant was noted, likely due slight difference between the mass of samples. Further variability may have been caused by the difficulty in ensuring representative samples due to the small pan size required. As mentioned in Chapter 2, Section 2.3.4 it is well known that specific heat capacity changes with temperature and moisture content. To ensure comparability between simulants, specific heat capacity was measured for dry simulant samples, thus eliminating any variation caused by differing moisture contents. Change in specific heat capacity with temperature was observed in the results for all the simulants tested at the University of Bath as shown in Figure 4-15. The simulants were split into two groups based on the average specific heat capacity measured at approximately 26.8°C to ensure clarity: Figure 4-15a shows simulants BM3, BM4, SE80, ND3 and ND4 whilst Figure 4-15b shows simulants SS9, BM1, BM2, BM5, BM6 and ND2. It was noted that the average specific heat capacity of simulants SS9, BM1, BM2, BM3, BM4, BM5, ND1 and ND2 reduced between 26.8°C and 36.5°C before increasing at all other temperatures consequently as shown in Figure 6-19. The behaviour of simulant BM6 was also unexpected. As shown in Figure 6-19a, the average specific heat capacity of simulant BM6 reduced between 26.8°C and 36.5°C, increased slightly between 46.3°C and 56.2°C and 56.2°C and 66.1°C then appeared to reduce. This was due to variability in the measurement of specific heat capacity between samples of simulant BM6. The median specific heat capacity values calculated for the simulants tested at the University of Bath are shown in Figure 4-16. From Figure 4-16a, it is clear that the median values for simulant BM6 follow the same pattern as that described for simulants SS9, BM1, BM2, BM3, BM4, BM5, ND1 and ND2.

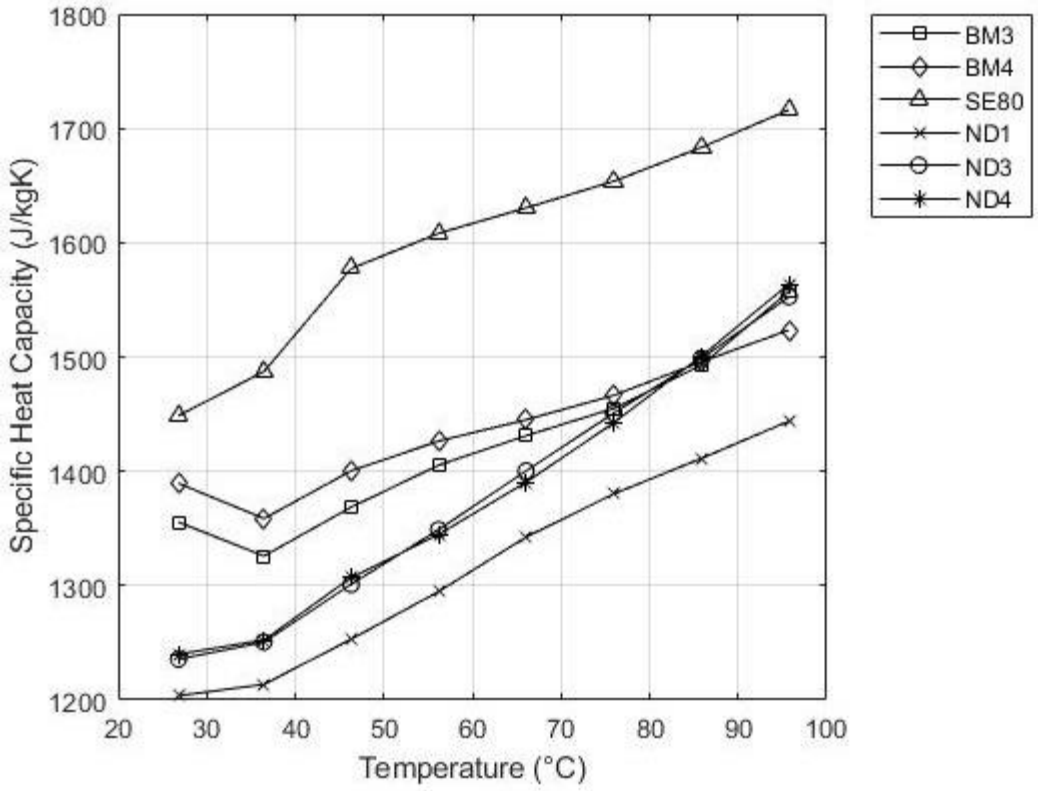


(a)

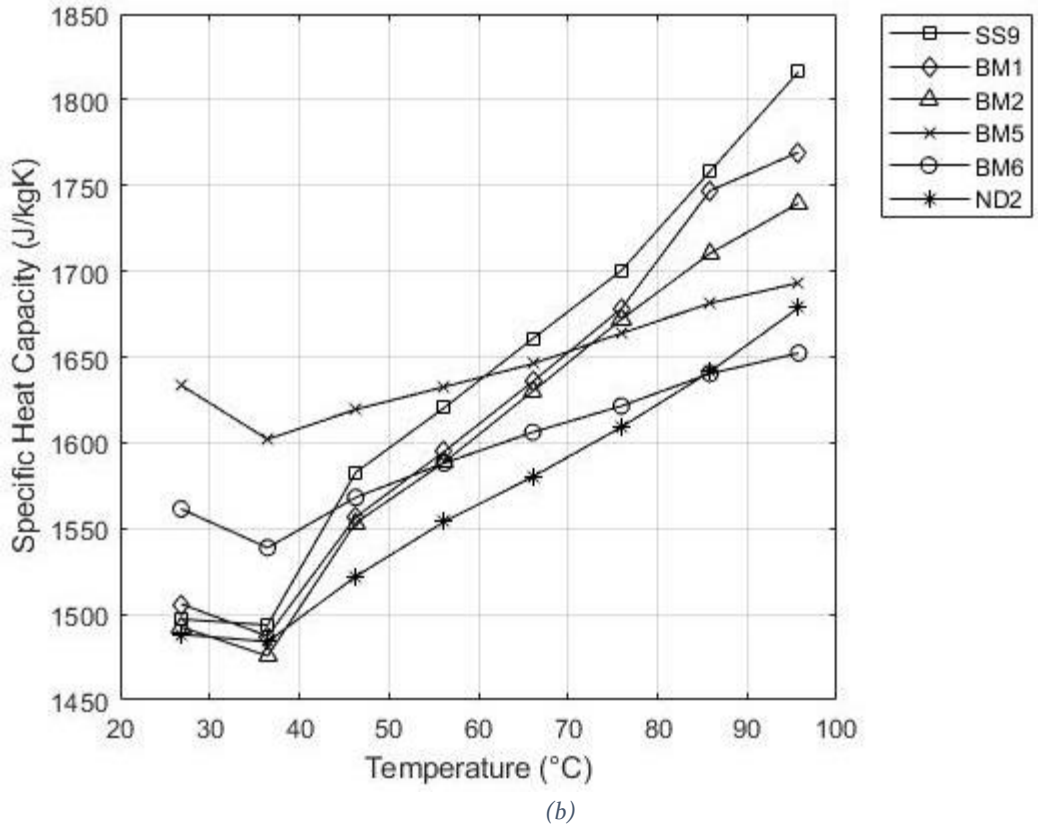


(b)

Figure 4-15. Average values of specific heat capacity (dry sludge) (J kg^{-1} K^{-1}) of faecal sludge simulants tested at the University of Bath at approximately 26.8°C, 36.5°C, 46.3°C, 56.2°C, 66.1°C, 78.0°C, 85.9°C and 95.7°C



(a)

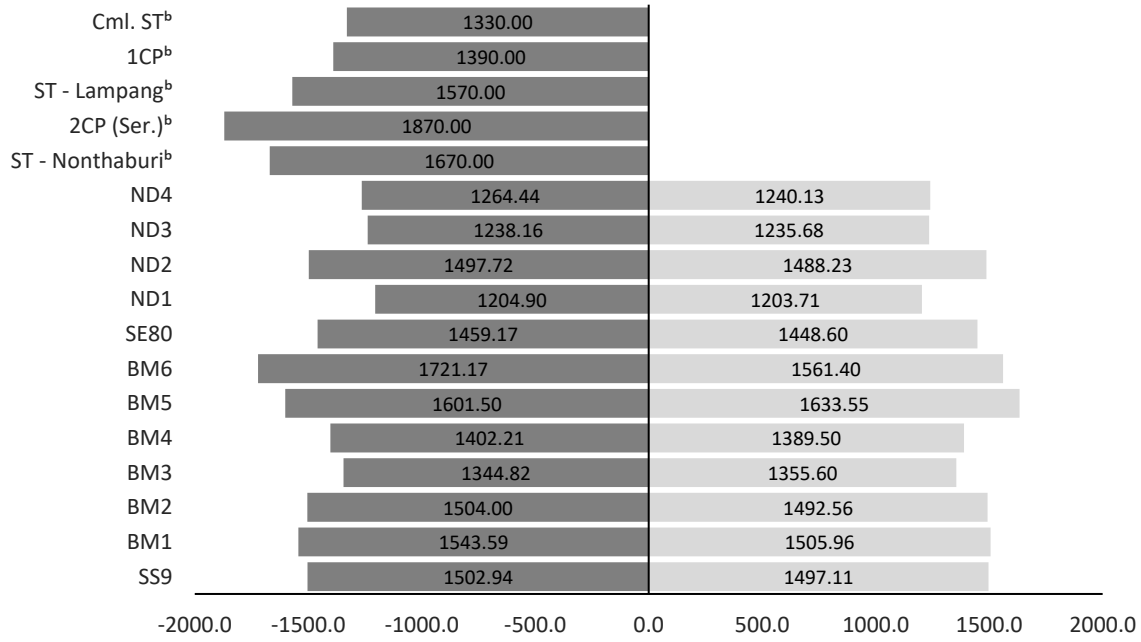


(b)

Figure 4-16. Median values of specific heat capacity (dry sludge) (J kg^{-1} K^{-1}) of faecal sludge simulants tested at the University of Bath at approximately 26.8°C, 36.5°C, 46.3°C, 56.2°C, 66.1°C, 78.0°C, 85.9°C and 95.7°C

Average and median values of specific heat capacity for the dry simulants tested at approximately 26.8°C are shown in Figure 4-17 alongside average values for real dry sludge at 25.0°C as reported by Koottatep *et al* (2012). Given the variability noted in the specific heat

capacity of the simulants, median values were considered to be more representative of the actual specific heat capacity of the simulants. Therefore, average values for real FS were compared with median values for the simulants. As shown in Figure 4-17, the median specific heat capacity for simulants SS9, BM1, BM2, BM6 and ND2 matched most closely with the average specific heat reported for sludge originating from a septic tank in Lampang. Simulant BM6 was the most representative with a percentage difference of -0.55%. Simulants BM3, ND1, ND3 and ND4 matched most closely to sludge originating from a commercial septic with percentage differences of 1.91%, -9.97%, -7.35% and -6.99% respectively. Of the remaining simulants BM4 and SE80 matched most closely with sludge originating from one cesspool and simulant BM5 matched most closely with sludge originating from a septic tank in Nonthaburi. The most representative simulant of real sludge was BM4, with a percentage difference of -0.04% between the median specific heat capacity and average specific heat capacity of sludge from one cesspool in series. Therefore, all of the simulant sludges appeared to compare well with the specific heat capacity of more stabilised sludge.

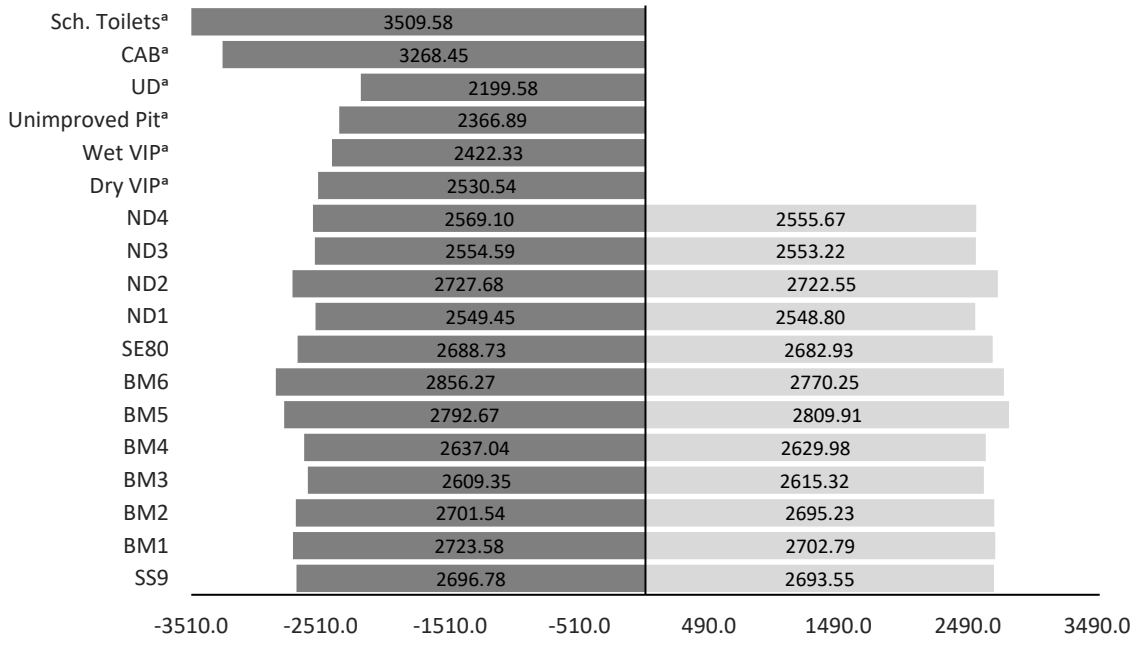


[b] Koottatep *et al* (2012)

Figure 4-17. Bi-directional chart showing average (left hand side) and median (right hand side) specific heat capacity (dry sludge) (J kg^{-1} K^{-1}) of real faecal sludge at approximately 25°C and simulants SS9, BM1-6, SE80 and ND1-4 at approximately 26.8°C where Cml. ST is commercial septic tank, 1CP is one cesspool, ST is septic tank and 2CP (Ser) is two cesspools in series

For comparison with real FS results reported by Velkushanova *et al* (2017), the specific heat capacity of dry simulant samples was converted to specific heat capacity of wet samples using

Equation 4.6 recommended by Vaxelaire and Puiggali (2002).[13] Average results of wet specific heat capacity for simulants tested at the University of Bath and real FS reported by Velkushanova *et al* (2017) are shown in Figure 4-18. The median specific heat capacities of simulants SS9, BM1-6, SE80 and ND1-4 were most comparable with the averages reported for specific heat capacity of real FS originating from dry and wet ventilated improved pit latrines. Based on percentage difference in average specific heat capacity between real FS and median specific heat capacity of the simulants tested at the University of Bath, the most representative simulant was ND1 (0.72%) for real FS from dry VIP latrines. Simulants ND3 and ND4 were also representative of real FS from dry VIP latrines with percentage differences of 0.89% and 0.99% respectively. The most comparable simulant to sludge from an unimproved pit i.e. largely stabilised sludge, was simulant ND1.



[a] Velkushanova *et al* (2017)

Figure 4-18. Bi-directional chart showing average (left hand side) and median (right hand side) specific heat capacity (wet sludge) (J kg^{-1} K^{-1}) of real faecal sludge at approximately 20.0°C and simulants SS9, BM1-6, SE80 and ND1-4 at approximately 26.8°C where Sch. Toilets is school toilets, CAB is community ablution block, UD is urine diversion and VIP is ventilated improved pit

The comparability of both dry and wet specific heat capacity results for the simulants tested at the University of Bath with values reported for real FS appear to suggest that determination of specific heat capacity is robust to both the method used and whether samples are dry or

[13] It was not possible to make the same calculation for the dry sludge samples reported by Koottattep *et al* (2012) as the moisture content of the samples was unknown.

wet. The same method was used to determine specific heat capacity of dry samples for simulants and real FS tested by Koottatep *et al* (2012). Using the technique recommended by Vaxelaire and Puiggali (2002) to give results for wet specific heat capacity for the simulants tested at the Univerisity of Bath comparted well with those reported by Velkushanova *et al* (2017) determined by a different method. This would appear to suggest that the technique recommended by Vaxelaire and Puiggali (2002) is successful in adjusting dry specific heat capacity to wet specific heat capacity. This is useful as determining the dry specific heat capacity of a sample may reduce variability due to moisture content. It may also allow quick estimation of the likely specific heat capacity with different moisture contents of a real FS if the dry specific heat capacity is known. However, it is dependent on the dry components of samples being representative, which is unlikely to be the case with FS, even if originating from the same containment technology. Therefore, it is more likely to be of use for estimating the likely specific heat capacity of simulants where the assumption that samples are representative is suitable.

Boxplots of the thermal conductivity of wet samples of simulants SS9, BM1-6, SE80 and ND1-4 are shown in Figure 4-19. Relatively little variation in results was observed across all of the simulants tested, with the least variation exhibited in results for simulants SS9, BM1, BM6 and SE80. Simulants ND3 and ND4 had the highest thermal conductivity of the simulants tested whilst simulant SS9 had the lowest. However, it was noted that the thermal conductivity of all of the simulants was very low and similar to that of air (0.025 W mK^{-1} at 20°C).

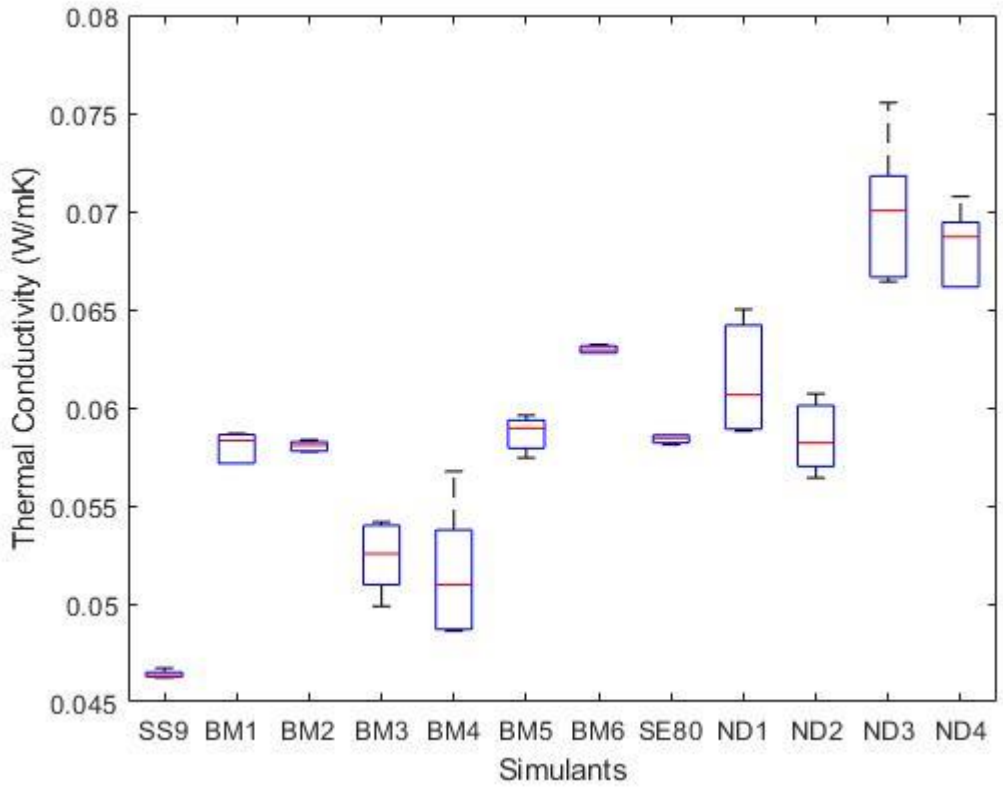


Figure 4-19. Boxplots for thermal conductivity (W/mK) of faecal sludge simulants SS9, BM1-6 and ND1-4 *tested at the University of Bath at approximately 20.0°C*

Prior to testing, every sample was mixed for approximately 3 minutes to eliminate as many air bubbles as possible; however, the presence of air in the mixes may have been a cause for the low thermal conductivity. This would only have been applicable to the more solid simulants i.e. SS9, BM1-6, SE80 and ND4. Difficulty in ensuring samples were maintained in suspension for simulants ND1-3 may also have caused issues with the probe. The TP02 needle probe was recommended to determine thermal conductivity based on available apparatus, time required for testing and its compatibility with slurries and pastes (Hukesflux Thermal Sensors, 2003). According to the manufacturer were convection in the medium occurs, the signal versus logarithmic time graph is non-linear. As shown in Figure 4-20 convection was observed in the simulant samples. However, the linear portion of the signal versus logarithmic time graph could still be selected for analysis in accordance with the manufacturer's instructions. Due to time and financial constraints it was not possible to locate alternative apparatus for measurement of thermal conductivity of the simulants.

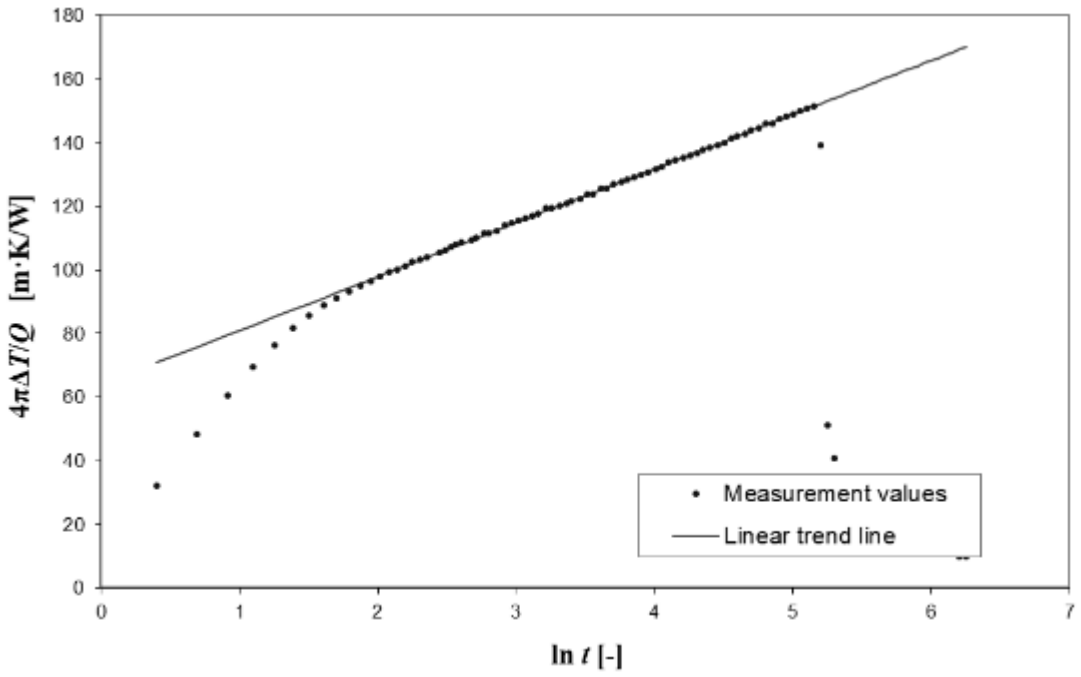
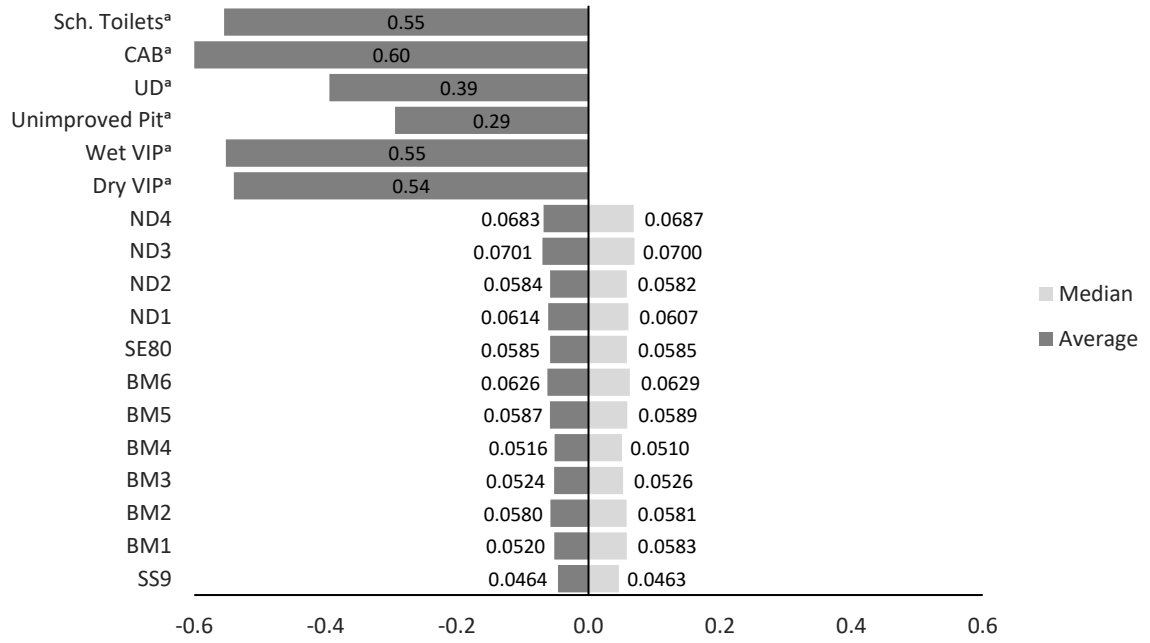


Figure 4-20. Signal (4πΔT/Q) versus logarithmic time (ln t) graph for measurement of thermal conductivity using thermal probe in simulant ND1 sample 1

As shown in Figure 4-21, the average thermal conductivity of the simulants tested at the University of Bath were smaller than those reported for real FS by a factor of approximately 100. The sample size of the sludge used at Bath was much bigger at 250ml compared to approximately 1.25ml (Velkushanova et al., 2017). It is likely that the difference was due to presence of trapped air in simulant samples or convection in the medium as discussed previously. Difference in methodology may also have been a factor. However, this is unlikely given that the results reported for dry FS by Koottatep *et al* (2012) and Makununika (2016) are very similar despite different methodologies, as shown in Table 4-10. It is interesting to note that the thermal conductivity values reported for real dry FS in Table 4-10 compare with the thermal conductivity values reported for the wet simulant samples. It is unlikely that this proves anything however and no conclusions could be drawn about the representativeness of the simulants in terms of thermal conductivity or the suitability of the TP02 needle probe.



[a] Velkushanova *et al* (2017)

Figure 4-21. Bi-directional chart showing average (left hand side) and median (right hand side) thermal conductivity (wet sludge) (W mK^{-1}) of real faecal sludge and simulants SS9, BM1-6, SE80 and ND1-4 (at approximately 20.0°C) where Sch. Toilets is school toilets, CAB is community ablution block, UD is urine diversion and VIP is ventilated improved pit

Table 4-10. Thermal conductivity (dry sludge) (W mK^{-1}) of real faecal sludge where ST is septic tank, 2CP (Ser.) is two cesspools in series, 1CP is one cesspool, Cml. ST is commercial septic tank and VIP is ventilated improved pit

Containment Technology/ Source	Thermal Conductivity (W mK^{-1})	Apparatus
ST[a]	0.04	PA Hilton B480 Thermal Conductivity Analyser (Koottatep et al 2012)
2CP (Ser.)[a]	0.04	
1CP[a]	0.05	
Cml. ST[a]	0.03	
VIP Sludge Pellets[b]	0.04	C-Therm TCi small volume test kit (Reddy, 2017)

[a] Koottatep *et al* (2012)
[b] Makununika (2016)

4.5.8 Most Representative Simulant

A number of characterisation tests were carried out on twelve simulants to assess which, if any, would be most representative of real type B FS. A summary of the characterisation tests and most representative simulant with regards to each sludge is shown in Table 4-11.

Representativeness was estimated based on percentage difference between median values (where reported) of real FS and median values (where applicable) of simulants tested. Real FS is arranged according to estimated stability based on the volatile solids to total solids ratio. For some real FS it was not possible to calculate the volatile solids to total solids ratio, so the location on the stability scale was estimated based on FS from similar containment technologies. Where this occurred, the relevant real FS is marked with an asterix (*). Particle size distribution and thermal conductivity are excluded from Table 4-11 as no conclusions could be drawn regarding the representativeness of any simulant tested at the University of Bath for real FS. Percentage difference is shown in brackets to indicate how closely the simulant matches the real FS for each characterisation test. Negative percentage differences indicate that the result of the characterisation test for the simulant is smaller than that reported for real FS for the same test. A positive value indicated the opposite. Boxes in Table 4-11 were left blank where no result for real FS was reported for that characterisation test

Table 4-11. Most representative simulant for real FS from a variety of containment technologies and sources where FF is fresh faeces, VIP is ventilated improved pit, CAB is community ablution block, UD is urine diversion toilet, Sch. Toilets is school toilets, Cml. ST is commercial septic tank, 2CP (Ser) is two cesspools in series, 1CP is one cesspool, ST is septic tank, Fl is fluidised, bott is bottom, w (%) is moisture content, VS (%) is volatile solids, TS (%) is total solids, ρ (kg m^3) is density, c_{dry} (J kg^{-1} K^{-1}) is specific heat capacity of dry sludge and c_{wet} (J kg^{-1} K^{-1}) is specific heat capacity of wet sludge

Degree of Stabilisation	Containment Technology/ Source	w (%)	VS:TS	VS (%)	Ash (%)	TS (%)	pH	ρ (kg m^3)	c_{dry} (J kg^{-1} K^{-1})	C_{wet} (J kg^{-1} K^{-1})
Raw	FF[a]	**BM3** (4.65%)	**BM3** (9.78%)	**BM4** (-2.76%)	**BM4** (16.69%)	**BM3** (-16.32%)				
Partially raw	Dry VIP[a]	**BM3** (0.48%)	**BM3** (35.77%)	**ND1** (24.08%)	**ND1** (-46.17%)	**BM3** (-1.85%)	**ND4** (-0.35%)	**BM6** (-0.28%)		**ND1** (0.72%)
	Wet VIP[a]	**BM3** (0.97%)	**BM3** (44.78%)	**ND1** (31.48%)	**ND1** (-55.05%)	**BM3** (-3.69%)	**ND4** (-8.15%)	**ND1** (-0.86%)		**ND1** (5.09%)
	CAB[a]	**BM3** (3.35%)	**BM3** (60.95%)	**ND1** (40.17%)	**ND1** (-63.68%)	**BM3** (-12.09%)	**ND4** (-6.17%)	**BM2** (0.22%)		**BM5** (-15.09%)
Partially stabilised	UD[a]	**BM3** (26.95%)	**BM3** (124.32%)	**ND1** (54.81%)	**ND1** (-50.04%)	**BM3** (-75.26%)	**ND4** (-3.30%)	**ND1** (-1.04%)		**ND1** (14.71%)
	Sch. Toilets[a]	**BM3** (23.80%)	**BM3** (126.43%)	**ND1** (74.71%)	**ND1** (-92.58%)	**BM3** (-58.61%)	**ND4** (-16.40%)	**ND3** (-0.57%)		**BM5** (-22.14%)
	ST[b]*							**ND2** (15.41%)	**BM5**[c] (-2.21%) **BM6**[d] (-0.55%)	
	ST - Bott. Sludge[b]	**ND1** (-12.97%)	**BM3** (134.08%)	**ND1** (195.97%)	**SE80** (191.57%)	**ND1** (172.96%)				
	1CP*[b]							**ND2** (14.52%)	**BM4** (-0.04%)	
	1CP - Fl. Sludge[b]	**ND1** (-9.94)	**BM3** (135.87%)	**ND1** (183.84%)	**SE80** (142.96%)	**ND1** (165.41%)				

	2CP (Ser.)*[b]							**ND2** (14.52%)	**BM5** (-13.50%)	
	2CP (Ser.) - Bott. Sludge[b]	**BM3** (91.29%)	**BM3** (136.75%)	**ND1** (196.81%)	**SE80** (185.20%)	**ND1** (177.30%)				
	1CP - Bott. Sludge[b]	**ND1** (-13.16%)	**BM3** (137.76%)	**ND1** (196.34%)	**SE80** (190.78%)	**ND1** (173.70%)				
Largely stabilised	ST - Fl. Sludge[b]	**ND1** (-10.73%)	**BM3** (138.40%)	**ND1** (187.52%)	**SE80** (168.60%)	**ND1** (120.38%)				
	2CP (Ser.) - Fl Sludge[b]	**BM3** (92.66%)	**BM3** (147.14%)	**ND1** (189.53%)	**SE80** (156.01%)	**ND1** (120.52%)				
	Unimproved Pit[aa]	**BM3** (62.75%)	**BM3** (156.91%)	**ND1** (89.26%)	**ND1** (-92.58%)	**BM3** (-96.54%)	**ND3** (-3.98%)	**ND2** (33.74%)		**ND1** (7.40%)
Stabilised	Cml. ST*[b]							**ND2** (15.86%)	**BM3** (1.91%)	
	Cml. ST - Fl. Sludge[b]	**BM3** (9.95%)	**BM3** (191.70%)	**ND1** (188.07%)	**ND4** (-4.35%)	**BM3** (-29.95%)				
	Cml. ST - Bott. Sludge[b]	**BM3** (4.90)	**BM3** (197.10%)	**ND1** (196.32%)	**ND3** (3.19%)	**BM3** (-13.96%)				

[a] Velkushanova et al (2017)
[b] Koottattep et al (2012)
[c] Simulant most representative of dry specific heat capacity of real FS originating from ST – Nonthaburi (Koottatep et al., 2012).
[d] Simulant most representative of dry specific heat capacity of real FS originating from ST – Lampang (Koottatep et al., 2012).

No single simulant was found be most representative of a specific FS for all of the properties tested. This was similar to the findings from comparisons of the simulants developed by Zuma (2013) and by PRG (2014). The majority of simulants found to be most representative of real FS were those developed at the University of Bath. However, significant differences were still observed between simulant results and real stabilised FS results in terms of organics content. The simulants tested were also found to have significantly higher total solids contents when compared to real partially stabilised, largely stabilised and stabilised FS. Consequently, the degree of stabilisation was also significantly higher for the simulants tested than for stabilised FS.

The aim of the characterisation tests was to find a simulant which best matched the properties of real type B FS found to affect dewatering and drying behaviour. Assuming that largely stabilised and stabilised sludge falls into the type B category the remainder of the discussion centres on these sludges. Simulant ND2 was found to be most representative of the density of real FS originating from an unimproved pit and a commercial septic tank. Simulant ND3 was most representative of the pH level of sludge originating from an unimproved pit. Simulant ND1 was found to be most representative of real fluidised sludge originating from a septic tank in terms of moisture content, total solids and volatile solids. However, the total solids and volatile solids contents of simulant ND1 were still significantly higher than for the real sludge. The same conclusion may be drawn for the representativeness of simulants SE80 (ash) and BM3 (volatile solids to total solids ratio) for fluidised sludge originating from a septic tank.

Simulant BM3 was also found to be most representative of the moisture content and volatile solids to total solids ratio of fluidised sludge originating from two cesspools in series. Simulant ND1 was most representative of the total solids and volatile solids content whilst simulant SE80 was found to be most representative of the ash content of fluidised sludge originating from two cesspools in series. However, the percentage differences shown in Table 4-11 indicate that the results for all the aforementioned simulants were much greater than for the real FS. Simulants BM3 (moisture content, total solids content and volatile solids to total solids ratio) and ND1 (volatile solids content, ash content and total carbon) were most representative of real FS originating from an unimproved pit. In keeping with previous observations, the results of the simulants were found to differ significantly to the results reported for FS from an unimproved pit for the same tests. Simulant ND1 was also found to be most representative of the wet specific heat capacity of sludge originating from an unimproved pit. As shown in Table 4-11, the percentage difference was low (7.40%) indicating a reasonable match. Simulant ND1 was also most representative of the total carbon of sludge

originating from a commercial septic tank. A close match was observed in the results for dry specific heat capacity between simulant BM3 and real FS originating from a commercial septic tank.

The moisture content and total solids content for simulant BM3 were found to be reasonably similar to the same properties for fluidised sludge and bottom sludge originating from a commercial septic tank. Simulant BM3 was also found to be most representative of fluidised sludge and bottom sludge originating from a commercial septic tank in terms of volatile solids to total solids ratio, although results for the simulant were still significantly higher than for the real FS. Simulant ND1 was most representative of the volatile solids content for fluidised sludge and bottom sludge originating from a commercial septic tank. As mentioned previously, the results for the simulant were significantly higher than for the real FS. Simulant ND4 was found to be representative of the ash content of fluidised sludge originating from a commercial septic tank whilst simulant ND3 was found to be most representative of bottom sludge, as shown in Table 4-11. ND1 was concluded to be the most successful simulant in terms of replicating the real FS in comparison with the other simulants tested; however, it remains significantly different to real stabilised FS in terms of key properties such as total solids, volatile solids and degree of stabilisation.

4.6 Summary and Conclusions

The characterisation of twelve FS simulants was described and discussed in this chapter. Seven simulants (SS9, BM1-6) were based on the simulant FS recipe developed by PRG (2014). Simulant SE80 developed by Penn *et al* (2018) was also analysed. A further four simulants were developed based on the original recipes developed by PRG (2014) and Penn *et al* (2018) as well as the work done by Radford (2011) and Zuma (2013). All simulants were characterised for: density; pH; moisture content; total solids; volatile solids; ash; particle size distribution; specific heat capacity and thermal conductivity. Methodologies for each test were recommended based on standard procedures for soils and wastewater sludges as well as an extensive literature review (see Appendix D).

It was not possible to draw any conclusions regarding thermal conductivity as the measurements recorded for the simulants tested at the University of Bath were lower than those reported for real FS by Velkushanova *et al* (2017) by a factor of 100. The thermal conductivities measured for the simulants were all similar to that of air. The presence of air bubbles in the simulants may have been a cause for the low thermal conductivity of simulants SS9, BM1-6, SE80 and ND4. Solid-liquid separation during the course of the test may have caused the low readings for simulants ND1-3.

Few conclusions could be drawn with regards to particle size distribution. It was not possible to perform dry sieve analysis with the simulants due to the manner in which they dried. The 300mm lens used in the Malvern Mastersizer X to analyse particle size distribution appeared to have a suitable range (1.2µm - 600µm) for determining the size distribution of the simulants. However, the lowest residuals recorded for simulants BM5, BM6, SE80 and ND1-4 indicated a poor fit of presentation data to measurement data. It was found that the majority of the particles in simulants SS9 and BM1-4 lay in the supra-colloidal range identified by Karr and Keinath (1978) as having the greatest influence on the rate of sludge dewatering. This would imply that these simulants are likely to be difficult to dewater.

Results showed that the original recipe developed by PRG (2014) was robust to changes in ingredients such as substituting active yeast for brewer's yeast or removing the yeast entirely or using different cellulose components. Bubbling was noted in simulants SS9, BM1 and BM2. The substitution of brewer's yeast for active yeast delayed bubble formation in simulants BM3 and BM4. The omission of yeast in simulants BM5, BM6 and ND4 and the use of yeast extract in simulants SE80 and ND1-3 resulted in no apparent bubbling. It was observed that all of the simulants tested at the University of Bath replicated the chemical characteristics of real FS. The simulants developed at the University of Bath (BM1-6 and ND1-4) were found to be more representative of real type B FS in terms of chemical properties and specific heat capacity. However, the organic matter content, total solids content and the degree of stabilisation was significantly higher for all simulants than for real type B FS. Based on the comparability of simulants with the properties of real FS, it would appear that one universal simulant which may replicate all of the key properties highlighted is unlikely.

The following chapter details the preliminary faecal sludge simulant drying tests undertaken to assess the behaviour of the simulants dried under constant conditions in terms of cracking, shrinkage and crust formation.

5 Preliminary Drying Tests for Faecal Sludge Simulants

This chapter details the preliminary drying tests undertaken to observe the appearance of shrinkage, cracking or crust formation in the developed FS simulants. Section 5.1 outlines the methodology employed for the drying tests and provides details of the climate chamber used for all of the drying experiments detailed in this chapter and Chapter 6

5.1 Methodology

An ACS Compact DY110 Test Chamber was used for all of the drying tests. The DY110 Test Chamber has a 110 L useful capacity, a temperature range between -70.0°C and +180.0°C and a humidity range of between 10.0% and 95.0%. For temperature control the chamber is equipped with a both a mechanical cooling system and a mechanical heating system. The cooling system ensures the interior of the chamber is cooled uniformly by means of the compression and subsequent expansion of a refrigerant gas while the heating system consists of electric heating elements located near the ventilation system so that the heated air is conveyed by means of ventilation throughout the interior of the test compartment (ACS, 2020b). In order to control the humidity, the chamber is equipped with an electric humidifier which injects steam through a hole in the air flow after the air recirculation fan ensuring aerosol-free humidification. Additionally, the humidifier is controlled by a dedicated algorithm to increase reliability. The interior compartment of the test chamber is dehumidified, by a mechanical system based on the "cold-finger principle". According to this principle, when an object at a lower temperature is exposed to a higher ambient temperature, the moisture in the air condenses on the surface of the cooler object. The element at the lowest temperature in the climate chamber is the evaporator and a dedicated section of this is used to dehumidify the test compartment when required (ACS, 2020b). An annotated picture of the external dimensions and key aspects of the chamber is shown in Figure 5-1. The internal set of the chamber during the drying tests is shown in Figure 5-2.

Figure 5-1. Annotated photograph showing external dimensions in mm and key aspects of ACS Compact DYII0 Test Chamber (ACS, 2020a)

Figure 5-2. Internal set up for faecal sludge simulant drying tests in ACS Compact DYI10 Test Chamber

The chamber was set to a temperature of approximately 20 °C ± 0.3°C and RH of approximately 60% ± 3%, based on the thirty year meteorology climatological averages for December in Bangladesh obtained from the NASA Langley Research Center (LaRC) POWER Project funded through the NASA Earth Science/Applied Science Program (NASA 2019). Only temperature and relative humidity can be controlled in the climate chamber, air flow rate could not be. This was not considered a serious limitation to the use of the climate chamber as a number of researchers found air flow rate did not have a significant impact on drying rates (Makununika 2016; Septien, Mirara, et al. 2018; Fumagalli and Freire 2007; Yaldýz and Ertekýn 2001).

Care was taken to ensure simulants were homogenised prior to sampling as detailed in Section 3.2 of Chapter 3. Three samples of each simulant of known initial moisture content – determined separately prior to the beginning of the experiment – were placed into glass beakers with sample diameter of approximately 60mm and sample thickness of approximately 15mm. These sample sizes are comparable to those used for other small scale sludge drying tests (Reyes et al., 2004; Septien, Mugauri, et al., 2018; Makununika, 2016;

Septien et al., 2017). Glass beakers were used so that it was clear what was occurring inside the sludge whilst it dried.

Each drying test lasted for a period of ten days as that is the length of time reported by Way (2013) to dewater a liquidy FS to a spadeable consistency suitable for co-composting. Ten days of dewatering on USDBs has also been reported by Cofie *et al* (2006) and Cofie and Koné (2009). To track the appearance of shrinkage, cracking and crust formation, samples were removed and photographed once per day over the course of the ten day drying test. Sludge samples were weighed at the beginning and end of each ten day drying test to ascertain the total mass loss over the drying period assuming that mass loss was equivalent to moisture loss.

5.2 Results and Discussion

Simulants SS9, BM1 and BM2 were the three simulants which contained active yeast and they behaved as expected. All three simulants developed bubbles and expanded quite rapidly between day one and day two as shown in Figure 5-3. A smooth skin was also noted to have formed by day two. As they dried the simulants were observed to become darker in colour and the smooth skin hardened to a thick, impermeable crust. Some shrinkage of the simulant samples was noticed following the collapse of the bubbles formed by the reaction of the yeast. As shown in Figure 5-3 the crust would often stay at the point in the beaker to which the simulant had expanded initially whilst the underlying sludge then collapsed. This behaviour is unlikely to be representative of the drying of real FS.

	Days		
	First	**Second**	**Last**
SS9			
BM1			
BM2			

Figure 5-3. Photographs showing bubble formation in samples of faecal sludge simulants SS9, BM1 and BM2 for the first, second and final days of a 10 day drying test in drying chamber under constant conditions of approximately 20°C and 60% RH

Simulants BM3 and BM4 also behaved as expected following observations during the simulant development as discussed in Chapter 3. Due to the substitution of brewers yeast for active yeast, bubble development was delayed in BM3 and BM4 to day four as shown in Figure 5-4. By day ten samples of simulants BM3 and BM4 had also developed a thick impermeable crust.

	Days		
	First	**Fourth**	**Last**
BM3			
BM4			

Figure 5-4. Photographs showing bubble formation in samples of faecal sludge simulatns BM3 and BM4, for first, fourth and final days of a 10 day drying test in drying chamber under constant conditions of approximately 20°C and 60% RH

No bubble formation was observed in samples of simulants BM5 and BM6 due to the absence of any yeast. Simulants SE80, ND1, ND2, ND3 and ND4 also did not contain any active yeast ingredients and, as expected, exhibited no signs of bubbling or expansion. Greater apparent shrinkage was observed in samples of simulant SE80 than in samples of simulants BM5, BM6 and ND1-4, as shown in Figure 5-5. As mentioned in Chapter 3, psyllium husk is known to contain mucilage which expands significantly on contact with water to form a mucilaginous mass (National Centre for Biotechnology Information, n.d.). Therefore, it stands to reason that psyllium husk would shrink significantly with the loss of water. Moisture loss could have been the cause of the more noticeable shrinkage during the drying tests in comparison with simulants SS9, BM1-6, and ND1-3. However, simulant ND4 contained a higher proportion of psyllium husk than simulant SE80 but it did not appear to shrink significantly more. This could imply that there are other factors in the sludge also responsible for determining amount of shrinkage.

According to Cai *et al* (2015) the addition of rice straw to sewage sludge cakes reduced the shrinkage of the cake by "forming a skeleton" for the sludge. This assumption is similar to that made by Ormeci and Vesilind (2000) regarding the design of a synthetic sludge recipe for activated sludge in which they included "filamentous microorganisms…to provide a backbone for the overall floc structure." Based on these assumptions, the addition of hemp fibre or compost into the sludge recipes may have acted as a skeleton thus limiting shrinkage

when they are present. However, no conclusion could be drawn as it was very difficult to ascertain the degree of shrinkage due to moisture loss from drying in simulants SS9 and BM1-4 because of the rise and fall of the sludges following the formation of bubbles.

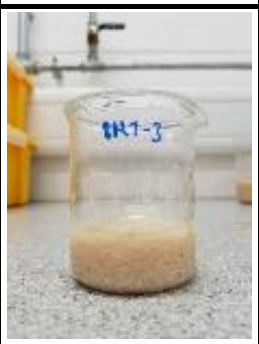
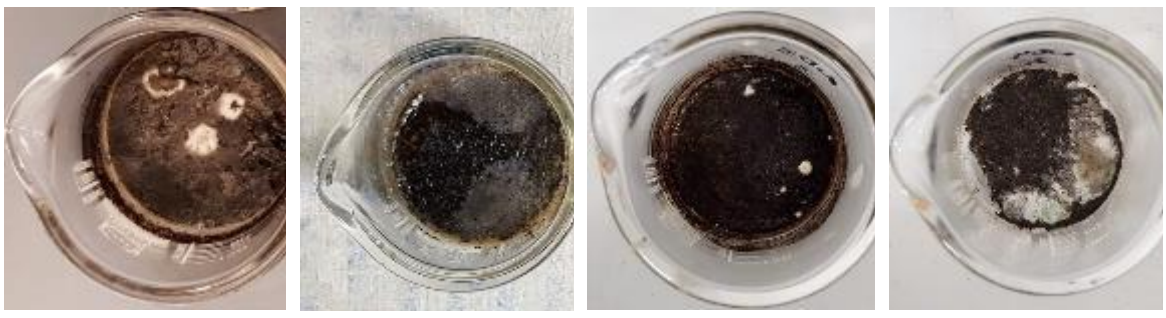
	Days			Days	
	First	Last		First	Last
BM5			ND1		
BM6			ND2		
SE80			ND3		
			ND4		

Figure 5-5. Photographs showing shrinkage of samples of synthetic FS simulants BM5 and BM6[14], SE80 and ND1-4 between day 1 and 10 of 10 day drying test in drying chamber under constant conditions of approximately 20°C and 60% RH

[14] BM5 and BM6 erroneously labelled as BM6 and BM7 respectively.

A thick impermeable crust was observed to have formed on the surface of samples of simulants BM5 and BM6 which had become very mouldy by the final day of the ten day drying test. In contrast, a smooth thin crust was observed on the surface of simulant SE80 samples on day three of the ten day drying test. By day ten no crust could be identified visually. On removal of the samples from the beakers it was noted that the simulants were brittle, almost crisp like, in comparison with other simulant samples tested. When samples of simulants SS9 and BM1-4 were removed from beakers they were often spongey. Simulants BM5 and BM6 were hard with a softer bottom.

A thin crust was observed on the surface of samples of ND1 and ND2 by day five and on ND3 by day seven. A thicker crust was observed on the surface of ND4, similar to that which formed on the surfaces of simulants SS9 and BM1-6, by day two. As shown in Figure 5-6 mould had formed on the surface of simulants ND1, ND3 and ND4 by the end of the ten day drying test. Based on visual observation, the crust formation on ND1-3 took more time in comparison with all other simulants. The surfaces of ND1-3 were observed to have a slight sheen as shown in Figure 5-6, in keeping with observations of real FS samples (Septien et al., 2017). Regarding the simulants, the sheen may have been due to the presence of oleic acid in the recipes as oleic acid is an unsaturated fatty acid which floats on water (National Centre for Biotechnology Information, n.d.). Given the greater apparent quantity of free water in simulants ND1-3 it could have been the case that some of the oleic acid present in the samples was not absorbed by the solid material. When the samples were removed from the beakers at the end of the ten day drying test, samples of simulants ND1-3 had to be scraped out in contrast with the other simulants (SS9, BM1-6, SE80) which came out largely whole. Samples of simulant ND4 were very similar to those of simulant SE80 as they were brittle and crisp like.



(a) *(b)* *(c)* *(d)*

Figure 5-6. Photographs showing the crust formed on the surface of samples of synthetic FS simulants (a) ND1, (b) ND2, (c) ND3 and (d) ND4 by day 10 of 10 day drying test in drying chamber under constant conditions of approximately 20°C and 60% RH

The total mass loss of the three samples for each simulant was averaged and is reported, alongside average initial moisture content, in Table 5-1. The lowest average mass loss was

recorded for simulant BM4 whilst the highest mass loss was recorded for simulant BM6 as shown in Table 5-1. However, simulant BM6 also had the highest initial moisture content of 86.16%. There were significant differences in the mass lost during drying despite similar initial moisture contents. For example, the average initial moisture content of BM5 was 86.24% and for BM6 it was 86.16% while the average mass loss for each was recorded as 28.03 g and 39.47 g respectively. This is a difference of almost 10.0 g despite very similar initial moisture contents. When the percentage decrease is considered the difference remains significant.

Table 5-1. Initial average moisture content (wet basis) ($w_{ave,i}$), average mass loss ($M_{ave,loss}$) and average percentage decrease of simulants dried over ten day period in drying chamber under constant conditions of approximately 20°C and 60% RH

Simulant	$W_{ave,I}$ (wb) (%)	$M_{ave,loss}$ (g)	% Decrease
SS9	78.93	28.16	70.28
BM1	80.11	22.89	55.55
BM2	81.30	27.89	68.58
BM3	80.37	22.67	51.93
BM4	79.95	21.53	48.80
BM5	86.24	28.03	68.83
BM6	86.16	39.47	75.83
SE80	79.95	35.70	73.32
ND1	82.52	32.90	73.93
ND2	81.29	36.76	79.08
ND3	85.26	34.48	81.88
ND4	84.15	32.77	77.14

The variation may have been caused by the overestimation or underestimation of initial moisture content as the moisture content of each mix was determined prior to drying due to the assumption of simulant homogeneity. Each sample or simulant may have had slightly different moisture distributions which may have caused the differences. As noted in the literature review in Section 2.1.1 the moisture distribution in sludges is affected by a variety of properties and the same or similar sludges are not guaranteed to have a similar moisture distribution.

Variation may also have been caused by variation in the initial mass of the simulants. The average mass loss of the three samples is reported instead of medians due to the small number of tests performed, thus it is more likely to be susceptible to extreme values. It is more likely that variations in the initial mass between samples is the cause of the discrepancy. The initial mass, final mass and total mass loss are shown for each sample of simulants BM5, BM6 and ND1-4 are shown in Table 5-2.

Table 5-2. Initial mass, (M_i), final mass (M_f) amd mass loss ($M_{ave,loss}$) for samples 1-3 of simulants BM5, BM6 and ND1-4 dried over ten day period in drying chamber under constant conditions of approximately 20°C and 60% RH

Simulant	Sample No.	M_i (g)	M_f (g)	Mass Loss
BM5	1	33.00	6.08	26.92
	2	46.76	22.31	24.45
	3	45.03	12.32	32.71
BM6	1	51.44	9.26	42.18
	2	51.11	10.77	40.34
	3	53.92	18.03	35.89
ND1	1	42.40	12.63	29.77
	2	44.01	10.38	33.63
	3	46.97	11.67	35.30
ND2	1	31.47	7.09	24.38
	2	45.55	9.77	35.78
	3	61.70	11.58	50.12
ND3	1	40.92	7.38	33.54
	2	42.08	8.00	34.08
	3	43.32	7.50	35.82
ND4	1	52.18	15.30	36.88
	2	41.33	8.65	32.68
	3	35.22	6.46	28.76

There is clearly some discrepancy in the initial mass between samples, while the sample masses for BM6, ND1 and ND3 are relatively similar, there are significant differences in the initial sample masses for BM5, ND2 and ND4. Given that the sample thickness and diameter were all the same, and evaporation could only occur from the surface of the simulant sludge, it is likely that the cause of the discrepancy shown in Table 5-2 was variations in initial mass of the individual samples. It is well known that the greater the total mass of an object the more energy can be stored in it thus increasing the heat requirement for evaporation which will slow the rate of evaporation down. Further lab tests in which the mass of the simulant sludges is varied would be required to provide conclusive proof that mass variation is the cause of the variation in moisture loss despite similar initial moisture contents.

5.3 Summary

Over the course of these preliminary drying tests the behaviour of the simulants during drying was monitored. Crusts were observed to form on the surface of most simulants in keeping with behaviour observed for real FS. Shrinkage was also observed in simulants BM5,

BM6, SE80 and ND1-4. Simulants PRG SS9 and BM1-4 expanded due to bubble formation and then appeared to shrink. No cracking was observed in the surface of any of the simulants in contrast to the expected behaviour of real FS dried on USDBs. On the basis of the characterisation tests and the preliminary drying tests, simulants ND1-4 were recommended for further drying tests to establish drying curves.

In the following chapter drying tests with UV radiation and a small-scale column drying test for faecal sludge simulants are analysed and described.

6 Faecal Sludge Simulant Drying Tests

As discussed in Chapter 2, Section evaporation from a sludge occurs in two or three stages (Gruter et al., 1990; Ferrasse et al., 2002; Flaga, 2005; Wang et al., 2007; Makununika, 2016). This chapter considers the development of sludge drying curves for the best performing FS simulants from the characterisation and preliminary drying tests. Drying tests with the addition of UV radiation were conducted in tandem with a small-scale column drying test using the same ACS Compact DY110 Test Chamber as used in the preliminary drying tests. The first section provides a discussion of models used for drying curves. The following section details the methodology for the drying tests with UV Radiation followed by a section on the methodology for the column drying test. The results of each test are then discussed and analysed separately in Sections 6.2 and 6.3.

6.1 Methodology

6.1.1 Drying of Faecal Sludge Simulants with UV Lamp

The main purpose of the drying tests with the addition of the UV lamp was to determine the impact of radiation on the extent of drying of the simulants and the formation of any crust. Septien *et al* (2018) noted that the impervious crust formed on real FS samples dried in a solar drier may have been hardened more through the combination of temperature and solar radiation. An OSRAM Ultra-Vitalux UV-A lamp was chosen for the drying test as it has been regularly used for sunlight stimulation, was relatively cheap and easy to fit into the climate chamber. The radiation lamp emits UVA, UVB and visible radiation emitting light "similar in its composition to the complete radiation mixture of natural alpine sunlight" (OSRAM, 2006).

To set up the lamp inside the drying chamber a special rig was constructed as shown in Figure 6-1. According to the manufacturer the most suitable distance between the material to be irradiated and the lamp is approximately 50cm. Due to the internal dimensions of the chamber this was not possible and the distance between the lamp and the samples was much less, at approximately 15cm.

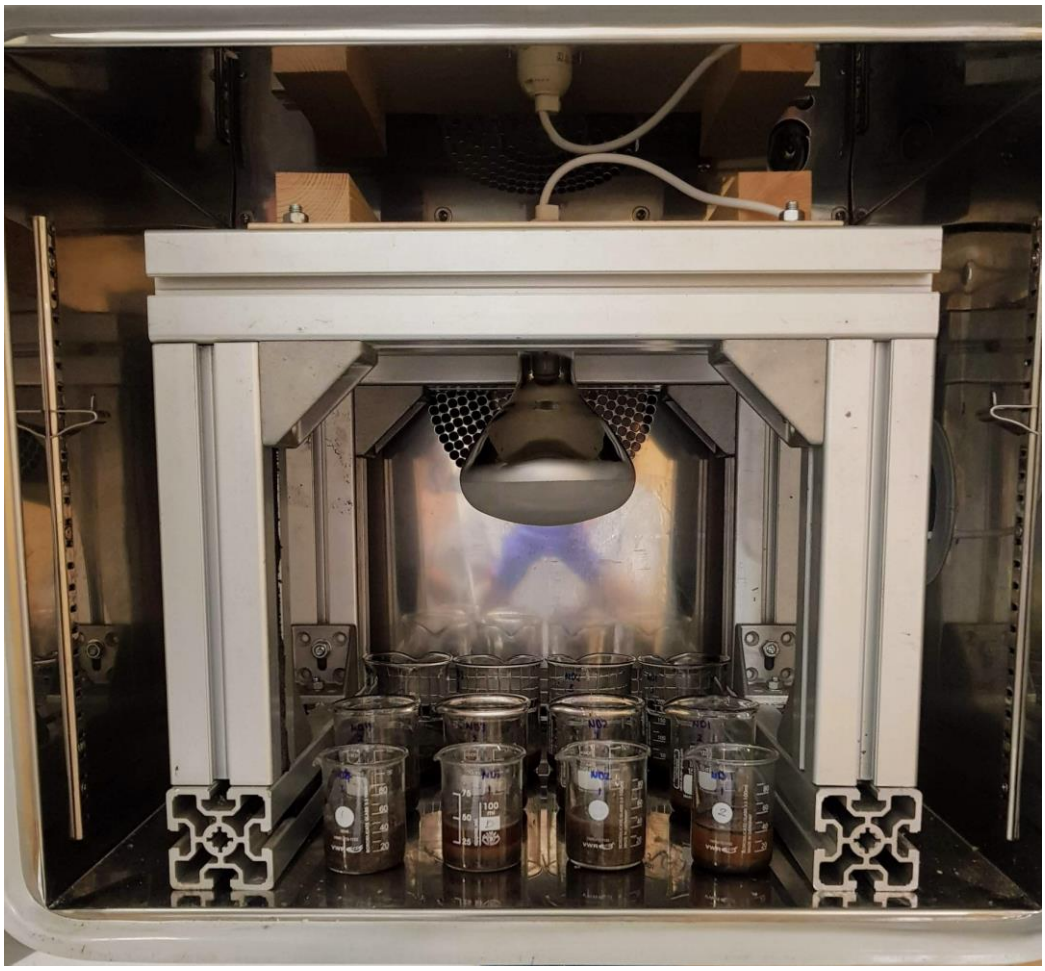


Figure 6-1. OSRAM Ultra-Vitalux UV-A lamp set up with faecal sludge simulants prior to beginning of drying test

The moisture content of each batch of recommended simulant was determined prior to the beginning of the ten day drying test. Three samples of each recommended simulant of known initial moisture content were tested. Three different sample sizes with diameters of approximately 50mm, 60mm and 70mm and thickness of 20mm, 15mm and 10mm respectively, were used. Each sample was removed from the chamber, weighed and photographed once a day over the ten day drying period to track crust formation and mass loss.

Mass loss during drying was used to calculate change in moisture content according to the procedure outlined in Mercer (2014). Moisture content was calculated on a dry basis as is customary (Hall, 1980). The drying rate, DR, was calculated according to:

$$DR = -\frac{dX}{dt}, \tag{6.1}$$

where dX/dt is change in moisture content, X (kg kg^{-1} dry basis), with time, t (s) (Hall, 1980; Reyes et al., 2004). Equation 6.1 was used to calculate drying rate due to the varying mass

transfer area of simulant samples during drying (Reyes et al., 2004). Mass ratio, MR_M, for each simulant sample was estimated as follows:

$$MR_M = \frac{M}{M_0}, \tag{6.2}$$

where M (g) is the sample mass at any instant and M_0 (g) is the initial mass of the sample. The simplified equations for estimating mass ratio were used due to the fluctuations in relative humidity during testing. This simplification is based on the influence of relative humidity in determining equilibrium moisture content.

Results of mass ratio versus time were fit to seven common semi-empirical thin layer drying models by non-linear multiple regression using Solver in Microsoft Excel 2016[15] to minimise the sum of square due to error in estimated mass ratio for each sample. The semi-empirical thin layer drying models identified in Chapter 4 were used for model fitting and are shown in Table 6-1.

Table 6-1. Semi-empirical thin-layer drying models where MR is moisture ratio or mass ratio, k, k_0, k_1 are drying constants, t (s) is time and a, b and c are empirical constants and n is the drying exponent (adapted from Celma et al (2012))

Model	Equation		Reference
Newton	$MR = exp(-kt)$	(6.3)	Lewis (1921)
Page	$MR = exp(-kt^n)$	(6.4)	Page (1949)
Henderson and Pabis	$MR = a\ exp(-kt)$	(6.5)	Celma *et al* (2012)
Cai	$MR = a\ exp(-(kt)^n) + b$	(6.6)	Cai *et a*l (2015)
Midilli-Kucuk	$MR = a\ exp(-kt^n) + bt$	(6.8)	Midilli *et al* (2002)
Logarithmic	$MR = a\ exp(-kt) + c$	(6.9)	Togrul and Pehlivan (2002)
Two term	$MR = a\ exp(-k_0 t) + \mathrm{b}\ \exp(-k_1 t)$	(6.10)	Henderson (1974)

The coefficient of determination (R^2), sum of squares due to error (SSE), root mean square error (RMSE) and reduced chi square (χ^2) were calculated for each model in order to test their accuracy in reproducing the experimental data. These parameters were calculated as follows:

$$R^2 = 1 - \left[\frac{\sum_{i=1}^{N}\left(MR_{exp,i} - MR_{pre,i}\right)^2}{\sum_{i=1}^{N}\left(MR_{exp,i} - \overline{MR}_{exp,i}\right)^2}\right], \tag{6.11}$$

$$SSE = \frac{\sum_{i=1}^{N}\left(MR_{exp,i} - MR_{pre,i}\right)^2}{N}, \tag{6.12}$$

[15] GRG (generalised reduced gradient) nonlinear multistart option was employed (Young, 2019).

$$RMSE = \left[\frac{1}{N}\sum_{i=1}^{N}\left(MR_{exp,i} - MR_{pre,i}\right)^2\right]^{1/2}, \quad (6.13)$$

$$\chi^2 = \frac{\sum_{i=1}^{N}\left(MR_{exp,i} - MR_{pre,i}\right)^2}{N - n}, \quad (6.14)$$

where $MR_{exp,I}$ is the experimental moisture ratio, $MR_{pre,I}$ is the predicted moisture ratio, $\overline{\mathrm{MR}}_{\mathrm{exp,I}}$ is the average experimental moisture ratio, n is the number of constants in the drying model and N is the number of data points (Celma et al., 2012; Kucuk et al., 2014; Makununika, 2016). An R^2 value close to 1.0 and low values of SSE, RMSE and χ^2 indicate a good fit to the experimental data.

6.1.2 Small-scale Column Drying Test

The primary aim of the small scale column drying test was to ascertain if a recommended simulant would dewater and dry in a similar way to real FS deposited on a covered USDB. When real FS is deposited on a USDB it is expected that 50-80% of the FS volume applied to the bed emerges as drained liquid (Heinss et al., 1998). It is also expected that sludge deposited on the bed develops a crust and cracks (Strande et al., 2014; Septien, Mugauri, et al., 2018; Way, 2013a). A successful simulant should exhibit similar behaviour. Finally, it was expected that the sludge would be of "spadeable" consistency at the end of the ten day drying test in keeping with observations made during the BM pilot project. The best performing simulant in terms of the characterisation tests and drying behaviour during the preliminary drying tests was recommended for testing.

A concrete UV lightbox[16] was used to house the drying column. The light box was connected to the chamber by way of a large pipe from the side and air with temperature and relative humidity of approximately 20°C and 60%, as used previously, was pumped into the box. Tiny tags were used to monitor the ambient temperature and RH within the box itself given the expectation it would vary slightly from the conditions within the climate chamber. The outer set up is shown in Figure 6-2.

[16] The concrete UV lightbox used in this experiment was designed for experiments for another project and retrofitted to suit the needs of the column drying test.

(a) (b)

Figure 6-2. Setup of UV concrete light box and climate chamber (a) light box with climate chamber (b) close-up of pipe connecting climate chamber and light box

A PVC pipe of 100mm diameter and 650mm height was fitted with a plastic bottom which had 2mm diameter holes drilled into it to allow any liquid run off. The empty PVC pipe was weighed and was then filled with approximately 100mm oven dried gravel (2 to 5mm diameter) and 100mm oven dried sand (0.2 to 0.6mm diameter) in keeping with the standard filter design of a USDB. A layer of coarser gravel was not used as its primary purpose is to serve as "base support for the filter medium (sand)" (Kuffour et al., 2009) and use would have increased the required length of PVC pipe beyond the capacity of the light box. Once the PVC pipe had been filled with the filter media, a hook was attached to it and the pipe was fitted to the underside of an electric balance which sat on the lid of the light box. The whole ensemble was then lifted above the box and lowered into it. The bottom of the pipe was placed over a plastic funnel (attached to the inner wall of the light box) which was connected by a hose to a sealed plastic container outside the light box as shown in Figure 6-3. Note that the scales and pipe were set to the back of the box due to the position of the pre-drilled holes in the box lid, but this was not considered an issue as the pipe itself was not sitting against the wall and therefore would not affect the mass measurement of the sludge.

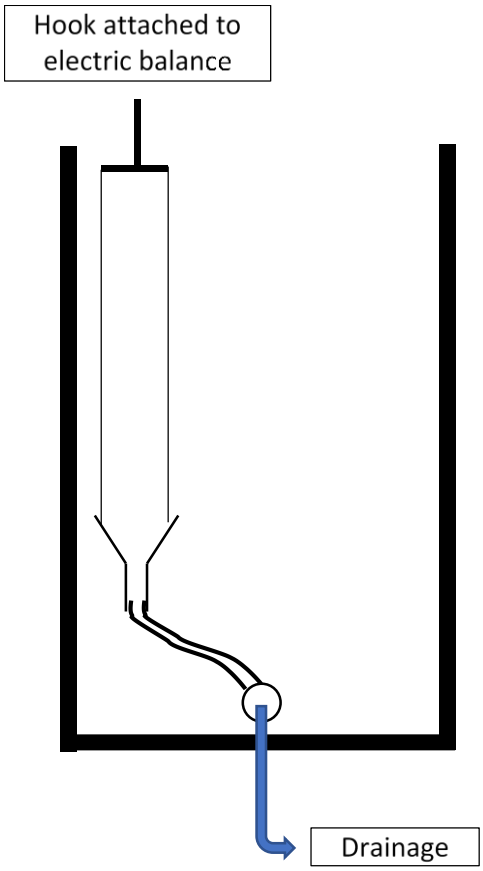


Figure 6-3. Schematic of UV light box with PVC pipe drying bed

The container to trap any runoff was checked observed for the first hour of testing and then checked once a day. Any runoff was recorded by emptying the container into a measuring cylinder and recording the liquid volume. This was similar to the method employed by Pescod (1971) when testing the drying of sludge on USDBs with sand filter. The PVC pipe was hung from a custom-made hook attached to the bottom of an electronic balance with a capacity of 0 to 30kg and an accuracy of ± 0.1g as shown in Figure 7-2. The balance was connected to a laptop to continuously record the weight of the sludge at 10 second intervals over the ten day drying period. Values per hour were averaged to obtain sample mass loss per hour accounting for moisture lost through drainage

After the pipe connecting the climate chamber to the light box was fitted, the apparatus was left for a two hour period to allow air from the climate chamber to circulate. The synthetic sludge was then poured into the pipe, on top of the filter media using a removable flexible pipe and funnel arrangement. This was to ensure as little disturbance to the filter as possible. The sludge used was poured in a layer of approximately 200mm as recommended by Pescod (1971) and used during the BH pilot study. During filling, six Thermochron 1-Wire DS1921G-F5 iButtons – similar to those used during the BH pilot study – were set to log temperature at 3 minute intervals and added to the sludge to determine temperature reached at the bottom of the simulant for comparison with results from the BH pilot study.

Experimental results were plotted as moisture content versus time, drying rate versus time and drying rate versus moisture content curves. Mass loss during drying was used to calculate change in moisture content according to the procedure outlined in Mercer (2014). The average drying rate per hour was calculated according to Equation 6.1.

6.2 Drying Tests with UV Radiation Results and Discussion

Prior to testing in the drying chamber with UV radiation the initial moisture content of each simulant was estimated by the method described in Chapter 6. Average and median values were calculated as shown in Table 6-2. The median initial moisture content was used in consequent calculations.

Table 6-2. Initial average moisture content (wet basis) ($w_{ave,i}$) and median moisture content (wet basis) ($w_{Q2,i}$) of simulants dried over ten day period in drying chamber with UV radiation

Simulant	$W_{ave,I}$ (wb) (%)	$W_{Q2,I}$ (wb) (%)
ND1	84.84	84.08
ND2	84.27	83.72
ND3	83.24	82.91
ND4	83.00	83.02

The primary purpose in the observation of the simulant samples during the ten day drying test was to monitor crust formation. Cracking and shrinkage were also noted where relevant. As expected, crusts appeared to form more quickly on the surfaces of all simulant samples than in the previous drying test without the addition of radiation. A thin skin was first observed on day two in most samples as shown in Figure 7-8. It was also observed that the surface of samples of simulants ND1-3 exhibited the same sheen as previously discussed in Section 7.4. As shown in Figure 7-8 this sheen reduced over the course of the drying test and the surface of most of the samples became duller. This is in keeping with observations made by Septien *et al* (2017) during solar drying of real FS. The crusts on ND1-3 were harder to poke through when removing samples than in the previous drying test without UV radiation. Whether the harder crust was due to the presence of the UV lamp or not could not be definitively determined as a result of this test.

Samples of simulant ND4 appeared to exhibit most shrinkage for possible reasons discussed in Section 5.2 . The shrinkage noted in simulants ND1-3 was predominantly in one direction (vertical) and uniform. Samples reduced in thickness during drying but retained the same diameter as the beaker in which they were tested. However, shrinkage in samples of simulant ND4 occurred in two directions (horizontal and vertical). As shown in Figure 6-4, samples of simulant ND4 were found to have come away from the edges of the beakers in which they

were tested. Shrinkage in the vertical direction was observed to have been non-uniform. The samples of simulant ND4 appeared to have 'puckered' and lifted up from the bottom of the beakers in which they were tested. Some apparent micro-cracks were observed on the crusts of samples of simulants ND1-3 by day nine of the ten day drying test but they appeared to be superficial. No cracks were observed in the surface of samples of ND4.

	Days			Days	
	Second	**Last**		**Second**	**Last**
ND1-1			**ND3-1**		
ND1-2			**ND3-2**		
ND1-3			**ND3-3**		
ND2-1			**ND4-1**		
ND2-2			**ND4-2**		
ND2-3			**ND4-3**		

Figure 6-4. Photographs of all samples of faecal sludge simulants ND1-4 on the second and the final day of the drying test in the climate chamber with UV radiation

A limitation to the test was the lack of data recorded for day two, day nine and day ten. On these days it was not possible to gain access to the lab to weigh or photograph the samples. Therefore, cubic spline interpolation function 'csaps' was used in Matlab to predict mass loss on the relevant days. This was necessary to allow comparison of the total mass loss of

simulants ND1-4 for drying with and without UV radiation. As shown in Table 6-3, the lowest mass loss for both drying tests was recorded for simulant ND4 whilst the highest was recorded for simulant ND2. As expected, greater mass loss was recorded for the simulants dried in the drying chamber with the addition of UV radiation. Using the interpolated data for day ten of the drying test with UV radiation there was a difference of 1.63g for simulant ND1, 0.85g for simulant ND2, 2.9g for simulant ND3 and 1.22g for simulant ND4. Using the recorded data for mass loss on day eight the differences were quite similar: 1.35g for simulant ND1, 0.86g for simulant ND2, 2.9g for simulant ND3, and 1.23g for simulant ND4. However, no conclusion could be drawn as to whether the increased mass loss was due radiation or slight increase in temperature or decrease in relative humidity, it could have been due to all three.

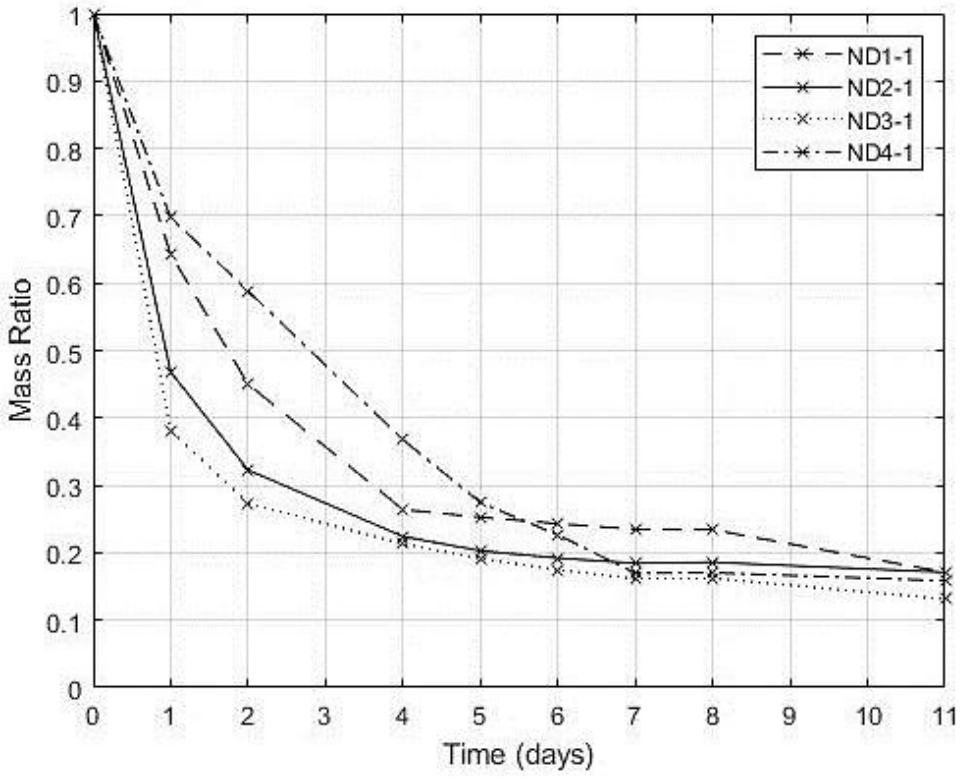
Table 6-3. Initial average moisture content (wet basis) ($w_{ave,i}$) and mass loss (M_{loss}) of simulants ND1-4 with sample thickness 15mm and diameter 60mm dried over ten day periods in drying chamber with and without UV radiation where interp. is interpolated

	Without UV Radiation		**With UV Radiation**		
		Day 10		**Day 8**	**Day 10 (interp.)**
Simulant	**$W_{ave,I}$ (wb) (%)**	**M_{loss} (g)**	**$W_{ave,I}$ (wb) (%)**	**M_{loss} (g)**	**M_{loss} (g)**
ND1	85.91	32.90	84.84	34.25	34.53
ND2	81.29	36.76	84.27	37.62	37.61
ND3	85.26	34.48	83.24	37.34	37.38
ND4	84.15	32.77	83.00	34.00	33.99

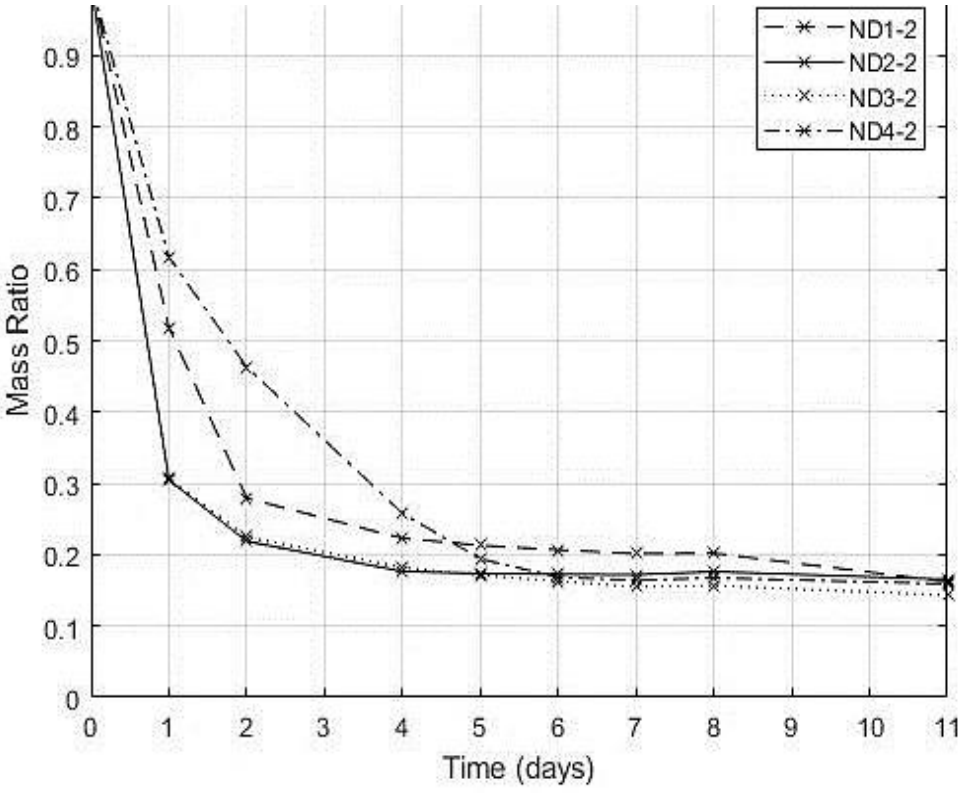
6.2.1 Drying Curves

Mass ratio versus time plots for samples 1, 2 and 3 of simulants ND1-4 are shown in Figure 6-5. The drying kinetics (behaviour) of each simulant tested was slightly different. As shown in Figure 6-5a, sample 1 of simulant ND4 exhibited the slowest reduction in mass ratio whilst simulants ND2 and ND3 exhibited relatively similar reductions. The mass ratio evolution of sample 1 of simulant ND1 was more rapid than for sample 1 of simulant ND4 but slower than for simulants ND2 and ND3. All of the simulant samples tended towards a similar final mass ratio as shown in Figure 6-5a. A similar trend was noted for sample 2 of each simulant as shown in Figure 6-5b. Sample 2 of simulants ND2 and ND3 exhibited a virtually identical evolution in mass ratio over the ten day drying test. Figure 6-5c shows the plot of mass ratio versus time for sample 3 of each simulant. The differences between the simulants tested appears to be is less than for the previous two sample sizes but simulant ND4 still exhibited the slowest evolution in mass ratio. Sample 3 of simulants ND1 and ND3 exhibited more similar evolution in mass ratio than simulants ND2 and ND3 in this case, as shown in Figure

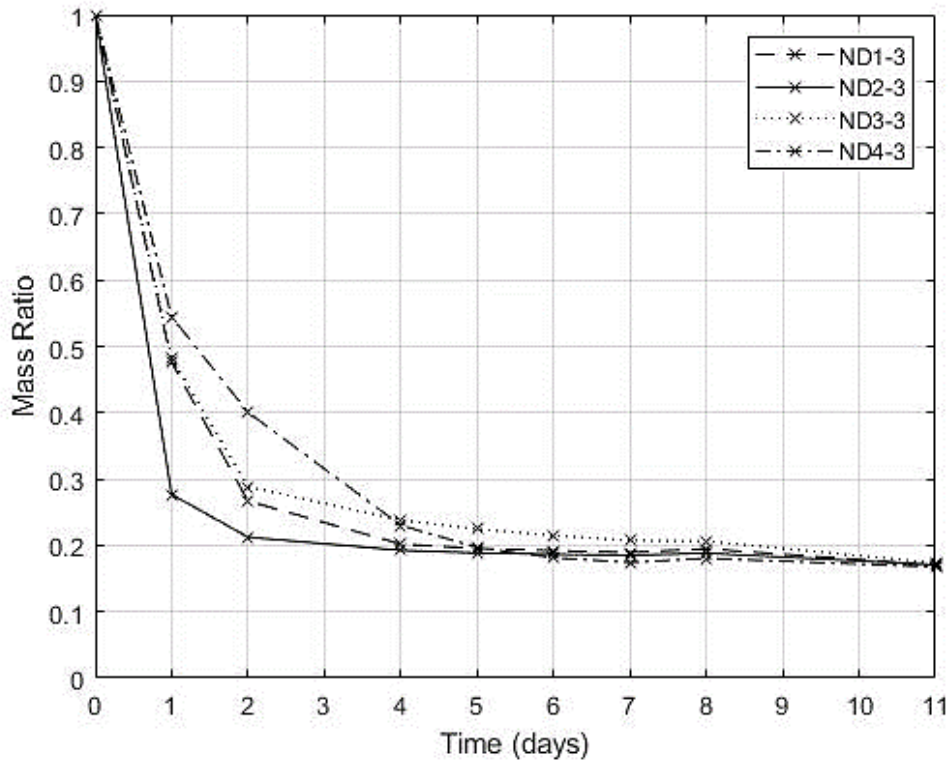
6-5c. It is likely that the mass ratio evolution of simulant ND4 was slower than the other simulants given the much higher proportion of psyllium husk present. As discussed previously, psyllium husk absorbs water to form a mucilaginous mass. It is likely that a greater proportion of water was absorbed and therefore more difficult to remove by drying resulting in slower loss of moisture in comparison with simulants ND1-3.



(a)



(b)



(c)

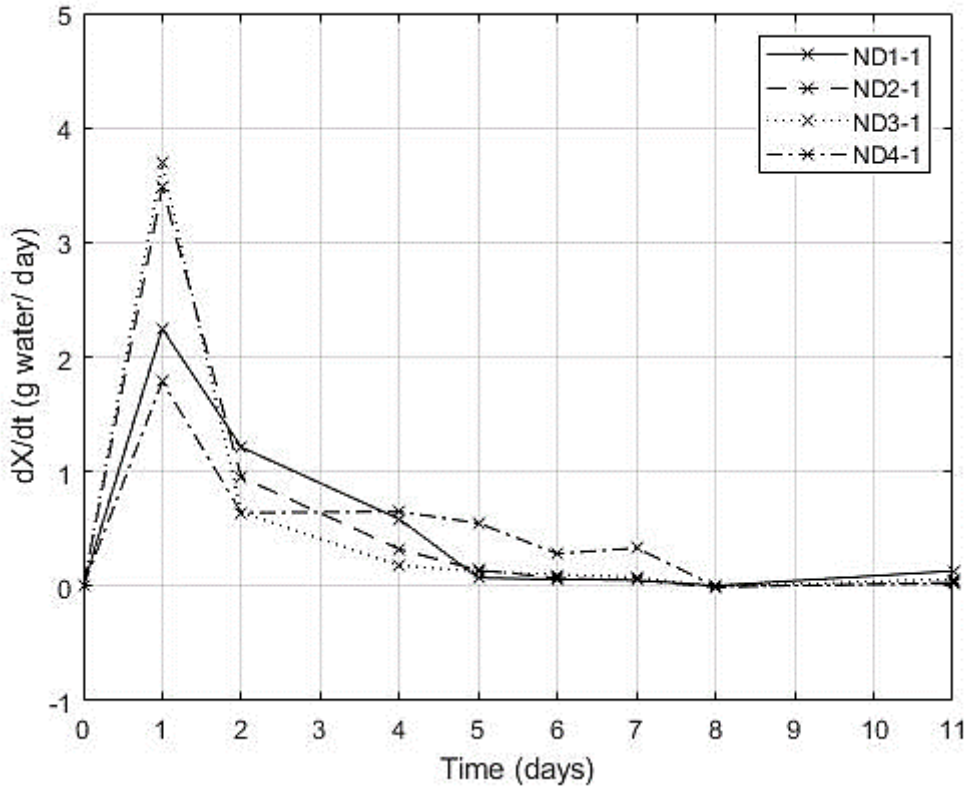
Figure 6-5. Mass ratio evolution as a function of time for simulants ND1-4 (a) sample 1, (b) sample 2 and (c) sample 3 where sample 1 had diameter of approximately 50mm and thickness 20mm, sample 2 had diameter of approximately 60mm and thickness of 15mm and sample 3 had diameter of approximately 70mm and thickness of 10mm

Drying rate versus time graphs were plotted for each simulant sample to identify the presence and duration of drying periods i.e. constant rate, first falling rate, second falling rate. The plots of drying rate versus time for samples 1, 2 and 3 of simulants ND1-4 are shown in Figure 7-9. A constant rate drying period could not be identified from any of the graphs. The constant rate period is the stage of drying during which free moisture is evaporated from the surface of the medium and is largely controlled by external conditions. Based on the evidence of drying of soils and sludges discussed in Chapter 4 it is likely that a constant rate period existed during the drying of the simulant samples. When the simulant samples were removed to be weighed and photographed on the second day of the drying test[17], the surface of most simulants was dry or had dry patches, as shown in Figure 7-8, indicating that the first falling rate period had begun (Inyang et al., 2018). However, due to the limited number of data points recorded for each sample over the ten day drying period it was not possible to ascertain for certain. The limited number of data points was justified in attempting to maintain constant conditions within the chamber by limiting the number of instances during which the UV lamp was switched off. A first falling rate period, followed by an apparent second falling rate period – during which drying continued but at a very low rate, could be identified

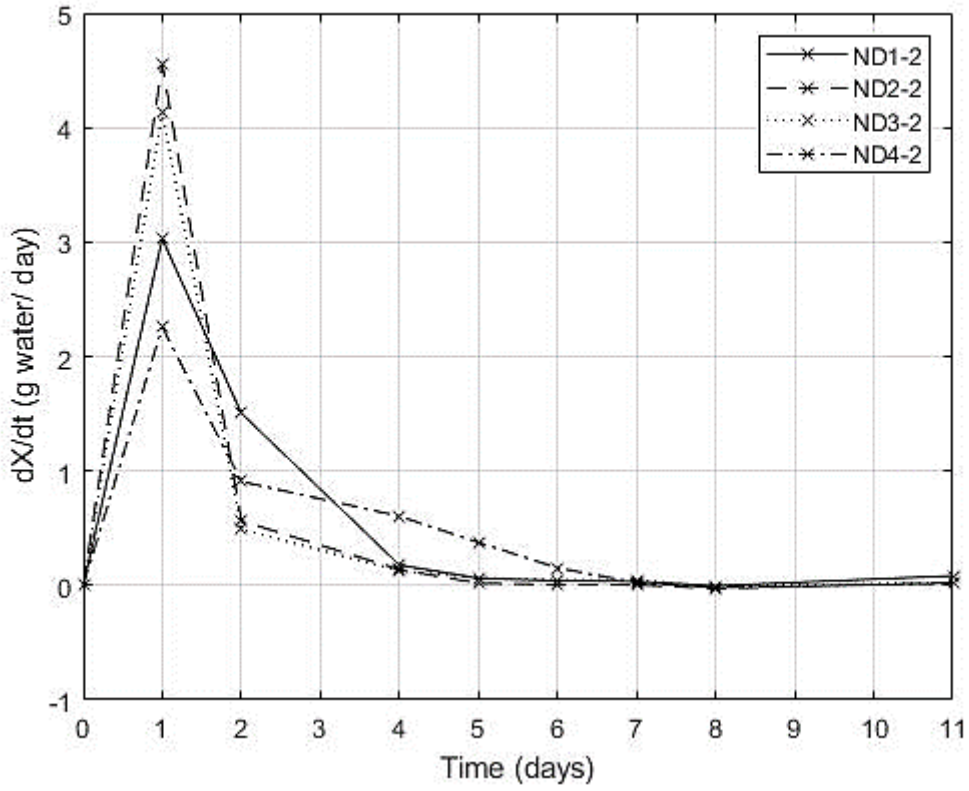
[17] The second day of the test is day one on the plots in Figure 6-5.

from the plots in Figure 7-9. The falling rate periods depend on the external conditions and the material transfer properties (Vaxelaire et al., 2000).

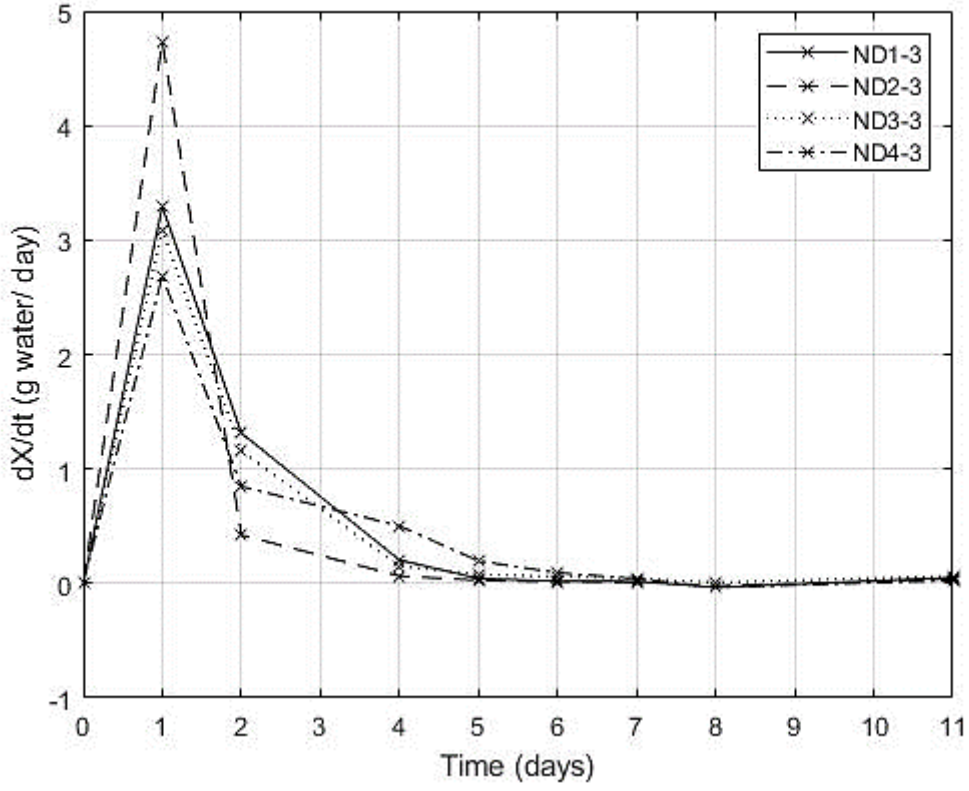
As shown in Figure 6-6a, the first falling rate period appeared to last to day 4 for simulants ND1-3. The second falling rate period was identified for each simulant sample as the point at which the graphs appeared to flatten as the equilibrium moisture content was approached (Smollen, 1990). For sample 1 of simulant ND4, the first falling rate period appeared to last until day 8 as shown in Figure 6-6a. According to Reyes et al (2014) denser, pasty sludges dry predominantly in the falling rate period whilst a long constant rate period is typically observed in the drying of diluted organic suspensions. Although no constant rate period was observed for simulants ND1-3 this may explain the reason for the initially lower drying rate and longer first falling rate period for all samples of simulant ND4. For sample 1 and 2, simulants ND2 and ND3 exhibited a similar evolution in drying rate with time; however, for sample 3, simulant ND2 exhibited a significantly higher initial drying rate than simulants ND1, ND3 and ND4 as shown in Figure 6-6c. The highest initial drying rates were observed for sample 3 of all of the simulants tested as shown in Figure 6-6. This was likely due to the larger surface area and smaller thickness of these samples. Despite the difference in drying rates between samples, the type of sludge appears to have been more significant in determining drying behaviour.



(a)



(b)



(c)

Figure 6-6. Drying rate as a function of time for simulants ND1-4 (a) sample 1, (b) sample 2 and (c) sample 3 where sample 1 had diameter of approximately 50mm and thickness 20mm, sample 2 had diameter of approximately 60mm and thickness of 15mm and sample 3 had diameter of approximately 70mm and thickness of 10mm

6.2.2 Thin-Layer Drying Models

The thin-layer drying models from Table 6-4 were fit to the experimental data using Solver in Excel 2016. As it was not possible to identify a constant rate period in the experimental data for any simulant tested, the models were fit to the data for the entire period of the drying test. The model constants for each model and the goodness of fit statistics are shown in Table 6-4. The models with the lowest R2 value and therefore the poorest fit for the experimental data for all simulant samples were the Newton and Henderson and Pabis models. The same models had the highest RMSE and SSE values. The best fitting models varied depending on the simulant and the sample. The Cai model was the best fit for samples 2 and 3 of simulant ND1, sample 1 and 2 of simulant ND2 and sample 2 of simulant ND3 with R^2 values of 0.996511, 0.999182, 0.999922, 0.999892 and 0.999880 respectively. Values of RMSE for each of the aforementioned simulant samples were 0.015102, 0.007325, 0.002240, 0.002668 and 0.002850 respectively. Values of SSE for each sample were 0.002053, 0.000483, 0.000045, 0.000064 and 0.000073 respectively. The sum of residuals was between -5.23×10^{-7} and 1.75×10^{-6}. Similar R^2 and RMSE values were reported for the Cai model when applied to the drying kinetics of sewage sludge (Cai et al., 2015). Based on the goodness of fit statistics, the best fit using the Cai model was achieved for ND2 sample 1 as shown in Figure 6-7b. The next best fit for samples 2 and 3 of simulant ND1, sample 2 of simulant ND2 and sample 2 of simulant ND3 was achieved using the Two Term model. However, the next best fit for sample 1 of simulant ND2 was achieved using the Midilli-Kucuk model.

The Midilli-Kucuk model was also the best fit for all samples of simulant ND4 with R^2 values of 0.994443, 0.998364 and 0.999037 for samples 1, 2 and 3 respectively. Values of RMSE for each sample were 0.020708, 0.011089 and 0.008154 respectively and values of SSE for each sample were 0.003860, 0.001107 and 0.000598 respectively. The sum of residuals was between -8.85×10^{-4} and 5.41×10^{-5}. The best fit of the model was achieved for simulant sample ND4-3 as shown more clearly in Figure 6-7d. The Two Term model was the next best fitting model for samples 1 and 2 of simulant ND4 whilst the Cai model achieved the best fit for sample 3 as shown in Table 6-4. The Two term model was the best fitting model for sample 1 of simulant ND1, sample 3 of simulant ND2 and samples 1 and 3 for simulant ND3 with R^2 values of 0.997154, 0.999863, 0.999554 and 0.998841 respectively. Values of RMSE for each of the simulant samples were 0.013685, 0.002964, 0.005448 and 0.008499 respectively. Values of SSE for each sample were 0.001685, 0.000079, 0.000267 and 0.000650 respectively. The sum of residuals was between -1.69×10^{-4} and 6.22×10^{-4}. The Cai model gave the next best fit for sample 1 of simulant ND1, sample 3 of simulant ND2 and samples 1 and 3 for simulant ND3 as shown in Table 7-5. The overall best fits were achieved for the data

recorded for simulants ND2 and ND3 as shown in Figure 6-7b and Figure 6-7c. However, there was relatively little difference in the goodness of fit statistics between the three best fitting models for all of the simulant samples, for example, there was a difference of 0.001325 between the R^2 values for the Two Term and Cai models calculated for sample 1 of simulant ND1. The worst fit was achieved for simulant ND4 using the Midilli-Kucuk model as shown in Figure 6-7d. There was greater variation between the experimental data and model data for all three samples of the simulant.

The Cai model was developed to model the drying kinetics of sewage sludge whilst the Two Term and Midilli-Kucuk models have previously been found successful for a number of food stuffs dried by solar drying or forced solar drying (Yaldýz and Ertekýn, 2001; Midilli, H. Kucuk, et al., 2002; Kucuk et al., 2014). Makununika (2016) found that the Page model best described the convective drying of FS pellets. Whilst it was not the best performing model, the Page model was found to fit the experimental data reasonably well with R^2 values above 0.97 and RMSE and SSE values below 0.39 and 0.013 respectively. However, results could not be compared as the conditions of the testing were not the same. Furthermore, the models describing the evolution of mass ratio for the real FS pellets appeared to have been applied to the data over the whole drying period despite the presence of a constant rate stage. As discussed in Chapter 2, Section 2.4 semi-empirical thin layer drying models such as those used in this thesis apply only to drying in the falling rate stage.

The Cai, Midilli-Kucuk and Two Term models were found to fit the experimental data well. However, due to time and financial constraints, it was not possible to develop the models to account for the influence of different factors affecting drying by running other drying tests. Therefore, the models cannot be used to predict mass loss under different conditions, limiting their applicability (Marzooghi and Dentel, 2014).

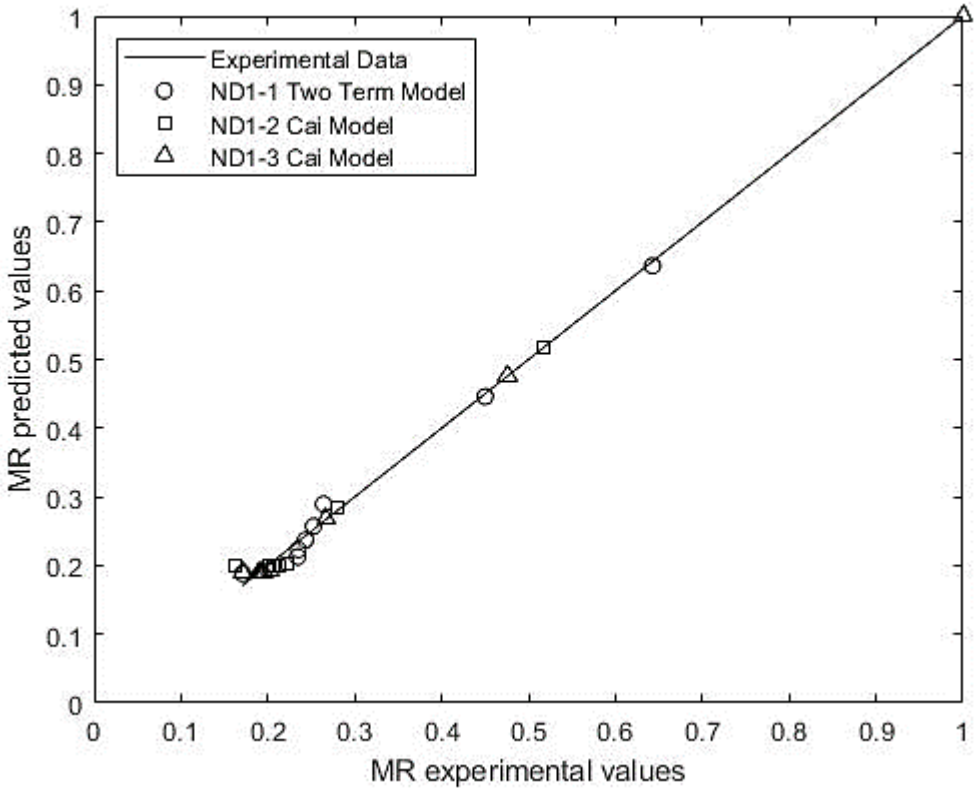
Table 6-4. Results of thin-layer drying models for each synthetic FS simulant sample for ten day drying test with UV lamp under average temperature and relative humidity condition of 22.0°C and 46.0%

Model	Sludge Sample	Model Constants	R^2	RMSE	SSE	Residuals
Newton (Lewis, 1921)	ND1-1	k = 0.275383	0.873060	0.091398	0.075182	7.62E-02
	ND1-2	k = 0.393638	0.761792	0.124780	0.140130	3.15E-01
	ND1-3	k = 0.451540	0.751020	0.127802	0.147000	4.33E-01
	ND2-1	k = 0.405548	0.769870	0.121943	0.133830	3.14E-01
	ND2-2	k = 0.837094	0.701698	0.140050	0.176526	8.78E-01
	ND2-3	k = 0.905996	0.637679	0.152329	0.208837	9.89E-01
	ND3-1	k = 0.543804	0.755874	0.127469	0.146236	5.16E-01
	ND3-2	k = 0.820210	0.744203	0.131337	0.155244	8.02E-01
	ND3-3	k = 0.384643	0.714300	0.133457	0.160298	3.03E-01
	ND4-1	k = 0.249987	0.975011	0.043912	0.017355	1.62E-02
	ND4-2	k = 0.323732	0.933226	0.070843	0.045169	1.22E-01
	ND4-3	k = 0.354959	0.866935	0.095836	0.082660	1.91E-01
Page (Page, 1949)	ND1-1	k = 0.520648 n = 0.550860	0.982779	0.033665	0.010200	1.96E-02
	ND1-2	k = 0.797459 n = 0.381542	0.977072	0.038713	0.013488	1.27E-02
	ND1-3	k = 0.880494 n = 0.343803	0.980104	0.036127	0.011746	1.18E-02
	ND2-1	k = 0.829908 n = 0.367163	0.992649	0.021794	0.004275	8.81E-03
	ND2-2	k = 1.272074 n = 0.172382	0.996183	0.015841	0.002259	1.82E-03
	ND2-3	k = 1.348416 n = 0.120373	0.998337	0.010319	0.000958	4.80E-04
	ND3-1	k = 1.001326 n = 0.304864	0.998880	0.008634	0.000671	2.18E-03
	ND3-2	k = 1.233338 n = 0.210103	0.998566	0.009832	0.000870	1.71E-03
	ND3-3	k = 0.841157 n = 0.335093	0.986380	0.029139	0.007642	7.43E-03
	ND4-1	k = 0.341327 n = 0.792082	0.990704	0.026783	0.006456	1.42E-02
	ND4-2	k = 0.516500 n = 0.639517	0.986338	0.032044	0.009241	2.95E-02
	ND4-3	k = 0.656200 n = 0.500719	0.988408	0.028286	0.007201	1.84E-02

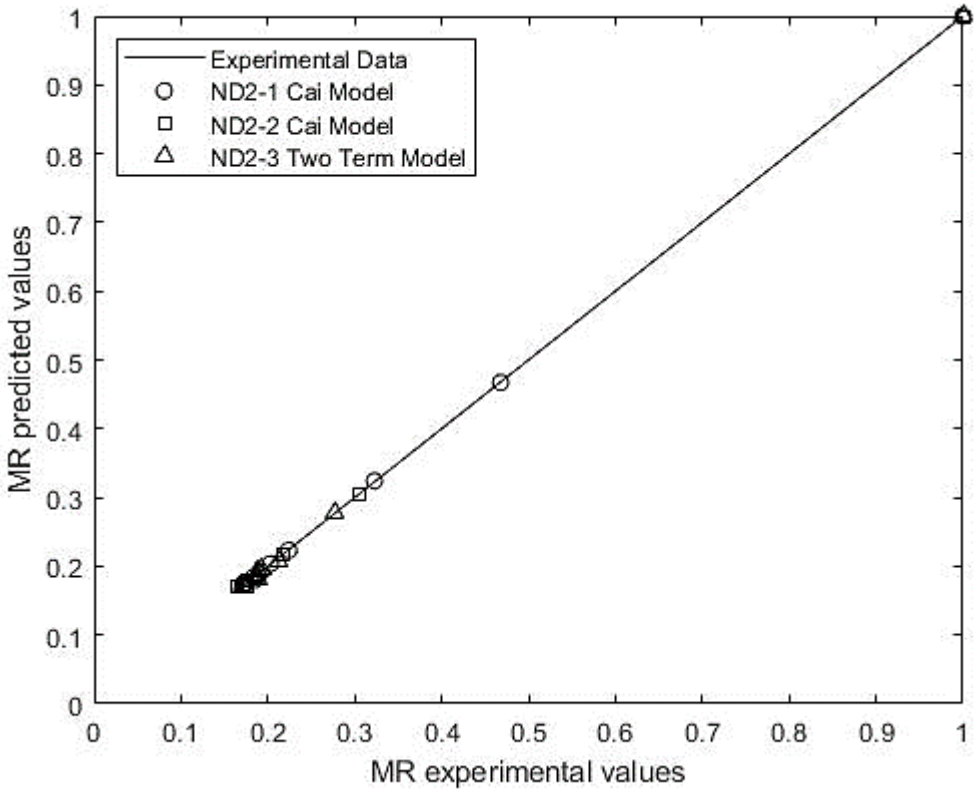
Model	Sludge Sample	Model Constants	R^2	RMSE	SSE	Residuals
Henderson and Pabis (Celma et al., 2012)	ND1-1	k = 0.237274 a= 0.891732	0.899324	1.085233	0.081395	0.059627
	ND1-2	k = 0.303736 a = 0.852789	0.799158	1.110771	0.114576	0.118149
	ND1-3	k = 0.340394 a = 0.856496	0.782143	1.156882	0.119547	0.128624
	ND2-1	k = 0.310722 a = 0.848433	0.809884	1.119527	0.110835	0.110560
	ND2-2	k = 0.747291 a = 0.946697	0.705247	1.535311	0.139214	0.174426
	ND2-3	k = 0.822038 a = 0.953688	0.640493	1.624279	0.151736	0.207215
	ND3-1	k = 0.388380 a = 0.852148	0.783715	1.158055	0.119981	0.129558
	ND3-2	k = 0.726374 a = 0.943211	0.748112	1.467361	0.130329	0.152872
	ND3-3	k = 0.281878 a = 0.827775	0.768694	1.069803	0.120083	0.129778
	ND4-1	k = 0.236686 a = 0.953548	0.979552	1.053934	0.039722	0.014201
	ND4-2	k = 0.294205 a = 0.925251	0.943631	1.116059	0.065090	0.038130
	ND4-3	k = 0.301472 a 0.887673	0.890480	1.131153	0.086944	0.068034
Cai (Cai et al., 2015)	ND1-1	k = 0.589699 n = 0.955805 a = 0.798303 b = 0.202593	0.995829	0.995827	0.000667	0.016567
	ND1-2	k = 0.942420 n = 1.289190 a = 0.799947 b = 0.200248	0.996511	0.996509	0.000686	0.015102
	ND1-3	k = 1.037592 n = 1.150269 a = 0.810727 b = 0.189361	0.999182	0.999181	0.000150	0.007325
	ND2-1	k = 1.044503 n = 0.716976 a = 0.827791 b = 0.172251	0.999922	0.999922	0.000016	0.002240
	ND2-2	k = 2.487720 n = 0.653564 a = 0.830236 b = 0.169759	0.999892	0.999890	0.000037	0.002668
	ND2-3	k = 4.186104 n = 0.539719 a = 0.818401 b = 0.181615	0.999648	0.999647	0.000123	0.004751
	ND3-1	k = 1.524211 n = 0.428323 a = 0.893553 b = 0.106536	0.999521	0.999517	0.000077	0.005645
	ND3-2	k = 3.054454 n = 0.450728 a = 0.859405 b = 0.140620	0.999880	0.999877	0.000020	0.002850
	ND3-3	k = 1.078303 n = 0.957192 a = 0.795119 b = 0.205363	0.996623	0.996621	0.000552	0.014509
	ND4-1	k = 0.330395 n = 0.946322 a = 0.889783 b = 0.103478	0.993637	0.993633	0.001140	0.022159
	ND4-2	k = 0.533855 n = 0.945775 a = 0.855016 b = 0.142395	0.997364	0.997364	0.000471	0.014076
	ND4-3	k = 0.707455 n = 0.829781 a = 0.840767 b = 0.158217	0.998569	0.998569	0.000247	0.009937

Model	Sludge Sample	Model Constants	R^2	RMSE	SSE	Residuals
Midilli-Kucuk (Midilli, H. Kucuk, et al., 2002)	ND1-1	k = 0.496206 n = 0.756205 a = 1.002944 b = 0.013857	0.992827	0.996238	0.021727	0.004248
	ND1-2	k = 0.765322 n = 0.603576 a = 1.002332 b = 0.014083	0.987194	0.991838	0.028932	0.007533
	ND1-3	k = 0.837456 n = 0.599788 a = 1.001554 b = 0.015249	0.992642	0.996876	0.021970	0.004344
	ND2-1	k = 0.801353 n = 0.547605 a = 1.000395 b = 0.011760	0.999318	1.000176	0.006637	0.000396
	ND2-2	k = 1.245125 n = 0.322069 a = 1.000091 b = 0.009825	0.999188	0.999655	0.007305	0.000480
	ND2-3	k = 1.336514 n = 0.199689 a = 1.000033 b = 0.005854	0.999114	0.999295	0.007531	0.000510
	ND3-1	k = 0.993584 n = 0.359119 a = 1.000124 b = 0.003829	0.999381	0.999658	0.006419	0.000371
	ND3-2	k = 0.993584 n = 0.359119 a = 1.000124 b = 0.003829	0.983939	0.992994	0.032910	0.009747
	ND3-3	k = 0.817129 n = 0.509422 a = 1.001191 b = 0.012063	0.992520	0.994844	0.021595	0.004197
	ND4-1	k = 0.317600 n = 0.932834 a = 0.992792 b = 0.008239	0.994443	0.992069	0.020708	0.003860
	ND4-2	k = 0.478497 n = 0.865278 a = 0.998155 b = 0.012513	0.998364	0.997429	0.011089	0.001107
	ND4-3	k = 0.619981 n = 0.716962 a = 0.999619 b = 0.012889	0.999037	0.999096	0.008154	0.000598
Logarithmic (Togrul and Pehlivan, 2002)	ND1-1	k = 0.594914 a = 0.791771 c = 0.207431	0.995730	0.995730	0.016763	0.002529
	ND1-2	k = 0.974122 a = 0.808542 c = 0.194511	0.995197	0.995195	0.017719	0.002826
	ND1-3	k = 1.069701 a = 0.814978 c = 0.186555	0.998768	0.998767	0.008989	0.000727
	ND2-1	k = 0.993948 a = 0.805413 c = 0.190390	0.996975	0.996973	0.013980	0.001759
	ND2-2	k = 1.796267 a = 0.824417 c = 0.175002	0.998975	0.998974	0.008208	0.000606
	ND2-3	k = 2.167666 a = 0.812657 c = 0.187149	0.999074	0.999072	0.007700	0.000534
	ND3-1	k = 1.271761 a = 0.822780 c = 0.173737	0.991152	0.991152	0.024267	0.005300
	ND3-2	k = 1.688749 a = 0.833861 c = 0.165003	0.996564	0.996563	0.015222	0.002086
	ND3-3	k = 1.069187 a = 0.793593 c = 0.206605	0.996602	0.996602	0.014555	0.001907
	ND4-1	k = 0.337647 a = 0.870011 c = 0.117356	0.993393	0.993393	0.022579	0.004588
	ND4-2	k = 0.537543 a = 0.844508 c = 0.149161	0.997124	0.997124	0.014703	0.001946
	ND4-3	k = 0.711805 a = 0.817667 c = 0.174368	0.996633	0.996633	0.015244	0.002091

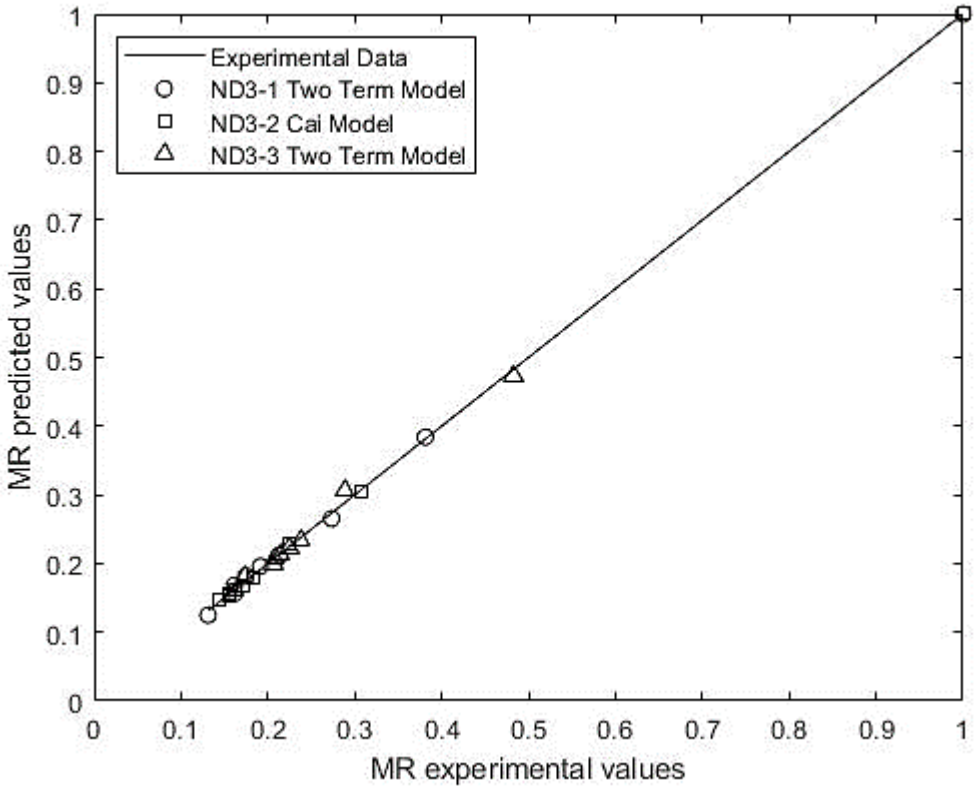
Model	Sludge Sample	Model Constants								R^2	RMSE	SSE	Residuals
Two term (Henderson, 1974)	ND1-1	k_0 =	0.035610	k_1 =	0.678202	a =	0.278243	b =	0.724636	0.997154	0.996932	0.013685	0.001685
	ND1-2	k_0 =	0.023549	k_1 =	1.050544	a =	0.229740	b =	0.774212	0.996218	0.996090	0.015722	0.002225
	ND1-3	k_0 =	0.009232	k_1 =	1.100123	a =	0.199023	b =	0.802839	0.998916	0.998903	0.008435	0.000640
	ND2-1	k_0 =	0.037694	k_1 =	1.144164	a =	0.248520	b =	0.749873	0.999054	0.999232	0.007819	0.000550
	ND2-2	k_0 =	0.016061	k_1 =	1.925974	a =	0.194346	b =	0.805378	0.999460	0.999482	0.005959	0.000320
	ND2-3	k_0 =	0.017646	k_1 =	2.400304	a =	0.209419	b =	0.790552	0.999863	0.999865	0.002964	0.000079
	ND3-1	k_0 =	0.074867	k_1 =	1.785805	a =	0.284702	b =	0.715069	0.999554	1.000174	0.005448	0.000267
	ND3-2	k_0 =	0.043510	k_1 =	2.049180	a =	0.218609	b =	0.781199	0.999496	0.999744	0.005830	0.000306
	ND3-3	k_0 =	0.031631	k_1 =	1.207499	a =	0.257159	b =	0.744406	0.998841	0.998701	0.008499	0.000650
	ND4-1	k_0 =	-0.175663	k_1 =	0.962601	a =	0.016380	b =	0.282140	0.994188	0.994934	0.021178	0.004037
	ND4-2	k_0 =	-0.019972	k_1 =	0.515178	a =	0.124659	b =	0.867302	0.997238	0.997246	0.014407	0.001868
	ND4-3	k_0 =	0.022297	k_1 =	0.760900	a =	0.207774	b =	0.786377	0.997003	0.997044	0.014383	0.001862


Experimental Data
ND1-1 Two Term Model
ND1-2 Cai Model
ND1-3 Cai Model
MR predicted values
MR experimental values
0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1

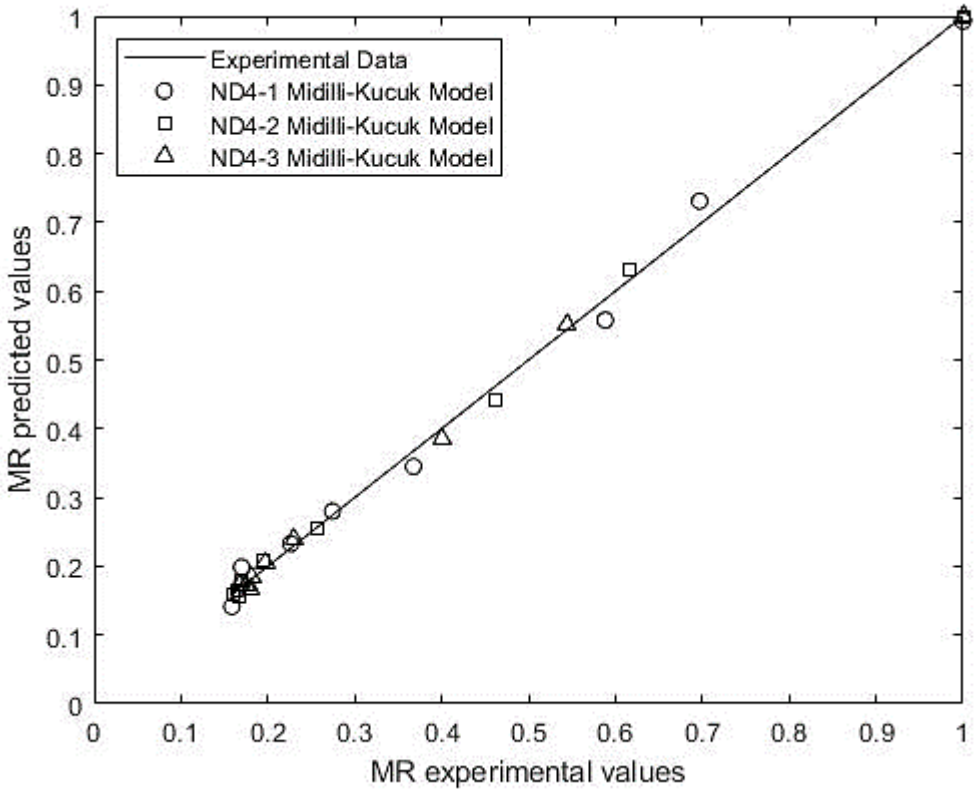
(a)


Experimental Data
ND2-1 Cai Model
ND2-2 Cai Model
ND2-3 Two Term Model
MR predicted values
MR experimental values
0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1

(b)



(c)



(d)

Figure 6-7. Best fit model for mass ratio versus time plots for faecal sludge simulants: (a) ND1; (b) ND2; (c) ND3; (d) ND4, where sample 1 had a diameter of approximately 50mm and thickness 20mm, sample 2 had diameter of approximately 60mm and thickness of 15mm and sample 3 had diameter of approximately 70mm and thickness of 10mm

6.3 Small-scale Column Drying Test Results and Discussion

A small-scale column drying test was run alongside the drying tests of simulant samples ND1-4 in the climate chamber with the addition of UV radiation. Based on the results of the characterisation tests discussed in Chapter 4 and the preliminary drying tests discussed in Chapter 5, simulant ND1 was recommended for testing in the column drying test. Simulant ND1 was found to be similar in consistency real FS as well as being most representative of key properties of largely stabilised and stabilised real FS out of the ND simulants, as shown in Table 6-5. A negative percentage difference is indicative of the result of the relevant test being lower for the simulant in comparison with the value reported for the same test for real FS. Where a box in Table 6-5 is blank it is because a value for that particular test was not reported for real FS and therefore could not be compared. The full comparison of simulants ND1-4 with results for real FS may be found in Appendix E.

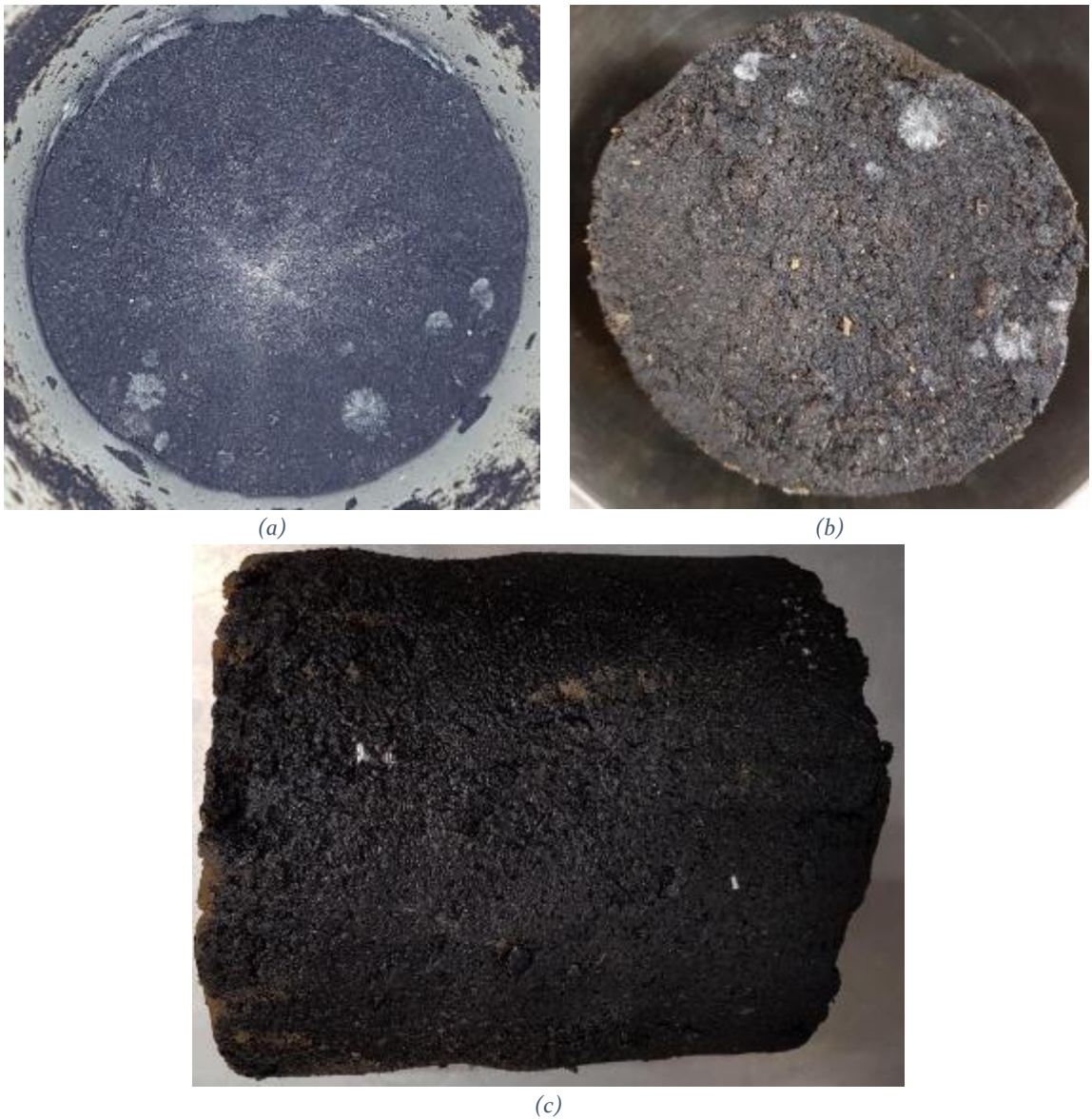
Table 6-5. Most representative simulant (ND1-4) for real faecal sludge from a variety of containment technologies and sources where UD is urine diversion toilet, Sch. Toilets is school toilets, Cml. ST is commercial septic tank, 2CP (Ser) is two cesspools in series, 1CP is one cesspool, ST is septic tank, Fl is fluidised, bott is bottom, ρ (kg m^3) is density, w (%) is moisture content, TS (%) is total solids, VS (%) is volatile solids, TC (%) is total carbon, c_{dry} (J kg^{-1} K^{-1}) is specific heat capacity of wet sludge, c_{wer} (J kg^{-1} K^{-1}) is specific heat capacity of dry sludge and NR is not reported

Degree of Stabilisation	Containment technology/ Source	ρ (kg m^3)	pH	w (%)	TS (%)	VS (%)	Ash (%)	VS:TS	TC (%)	c_{dry} (J kg^{-1} K^{-1})	C_{wet} (J kg^{-1} K^{-1})
Partially stabilised	UD[a]	**ND1** (-1.04%)	**ND4** (-3.30%)	**ND4** (31.19%)	**ND4** (-90.95%)	**ND1** (54.81%)	**ND1** (-50.04%)	**ND4** (131.67%)	**ND1** (54.81%)		**ND1** (14.71%)
	Sch. Toilets[a]	**ND3** (-0.57%)	**ND4** (-16.40%)	**ND4** (28.05%)	**ND4** (-75.45%)	**ND1** (74.71%)	**ND1** (-92.58%)	**ND4** (133.62%)	**ND1** (74.71%)		**ND2** (-25.26%)
	ST[b]	**ND2** (15.41%)							**ND1** (28.83%)	**ND2**[c] (-11.51%) **ND2**[d] (-5.35%)	
	ST - Bott. Sludge[b]			**ND1** (-12.97%)	**ND1** (172.96%)	**ND1** (195.97%)	**ND2** (196.28%)	**ND4** (140.66%)			
	1CP[b]	**ND2** (14.52%)							**ND1** (10.37%)	**ND2** (6.83%)	
	1CP - Fl. Sludge[b]			**ND1** (-9.94)	**ND1** (165.41%)	**ND1** (183.84%)	**ND2** (172.94%)	**ND4** (142.30%)			
	2CP (Ser.)[b]	**ND2** (14.52%)							**ND1** (12.18%)	**ND2** (22.74%)	
	2CP (Ser.) - Bott. Sludge[b]			**ND4** (94.68%)	**ND1** (177.30%)	**ND1** (196.81%)	**ND2** (193.41%)	**ND4** (143.11%)			
	1CP - Bott. Sludge[b]			**ND1** (-13.16%)	**ND1** (173.70%)	**ND1** (196.34%)	**ND2** (195.93%)	**ND4** (144.04%)			

Degree of Stabilisation	Containment technology/ Source	ρ (kg m^3)	pH	w (%)	TS (%)	VS (%)	Ash (%)	VS:TS	TC (%)	c_{dry} (J kg^{-1} K^{-1})	C_{wet} (J kg^{-1} K^{-1})
Largely stabilised	ST - Fl. Sludge[b]			**ND1** (-10.73%)	**ND1** (120.38%)	**ND1** (187.52%)	**ND2** (185.67%)	**ND4** (144.62%)			
	2CP (Ser.) - Fl Sludge[b]			**ND4** (96.02%)	**ND1** (120.52%)	**ND1** (189.53%)	**ND2** (179.54%)	**ND4** (152.60%)			
	Unimproved Pit[a]	**ND2** (33.74%)	**ND3** (-3.98)	**ND2** (66.80%)	**ND4** (-110.43%)	**ND1** (89.26%)	**ND1** (-92.58%)	**ND4** (161.47%)	**ND1** (89.26%)		**ND1** (7.40%)
Stabilised	Cml. ST[b]	**ND2** (15.86%)							**ND1** (24.68%)	**ND4** (6.99%)	
	Cml. ST - Fl. Sludge[b]			**ND4** (13.87%)	**ND4** (45.31%)	**ND1** (188.07%)	**ND4** (-4.35%)	**ND4** (192.66%)			
	Cml. ST - Bott. Sludge[b]			**ND4** (9.22%)	**ND4** (32.68%)	**ND1** (196.32%)	**ND3** (3.19%)	**ND4** (197.44%)			

[a] Velkushanova *et al* (2017)
[b] Koottattep *et a*l (2012)
[c] Simulant most representative of dry specific heat capacity of real FS originating from ST – Nonthaburi (Koottatep et al., 2012).
[d] Simulant most representative of dry specific heat capacity of real FS originating from ST – Lampang (Koottatep et al., 2012).

The median initial moisture content of simulant ND1 used in the column drying test was found to be 59.89% (wet basis). This was significantly lower than any previous batch of ND1 simulant and was likely a result of an error in the scaling up of the recipe for use in the column drying test. Due to time and financial resources it was not possible to repeat the drying test to account for the error. At the end of the ten day drying test the simulant was easily removed from the PVC column and maintained its cylindrical shape as shown in Figure 6-8. Some horizontal shrinkage of the simulant was noted in situ as shown in Figure 6-8a. Upon removal of simulant ND1 from the PVC drying column, a thin crust and some apparent desiccation of the surface was noted as shown in Figure 6-8b.



(a) *(b)*

(c)

Figure 6-8. Photographs of faecal sludge simulant ND1 following ten day small-scale column drying test (a) simulant sludge in situ in PVC drying column, (b) top surface of simulant sludge following removal from drying column, (c) cylinder of simulant sludge following removal from drying column

The final median moisture contents of the top, middle and bottom thirds of the simulant sludge were 44.76% (wet basis), 49.52% (wet basis) and 50.43% (wet basis) respectively. This would seem to indicate that the simulant was drier at the top. It may be said that the simulant sludge was dried from liquidy to spadable consistency within ten days, in keeping with observations made during the BH pilot test of covered USDBs in Bangladesh. On removal of the simulant sludge from the PVC drying column, electric veneer callipers were used to measure the dimensions and the final volume for the sludge was determined to be approximately 690.0cm^3. Assuming an approximate initial volume of 1570cm^3, the total percentage decrease of the simulant sludge over the ten day drying test was approximately 56.1%.

No runoff was observed over the duration of the drying test therefore implying that evaporation was the significant drying process in this case. The lack of observed runoff may have been due to the low moisture content and higher total solids of the simulant sludge. The initially bone dry sand and gravel filter media was found to be damp at the end of the ten day drying test, indicating some moisture had drained from the sludge. The moisture content of the filter media following the ten day drying test was found to be 8.30% (wet basis) and the total mass of moisture held in the filter media was calculated as 190.16g. It was not possible to ascertain the time frame over which the moisture was drained from the sludge. Based on an initial mass of 1582.00g, the percentage of water drained from the sludge over the course of the ten day drying test was found to be 12.02%. This is significantly lower than the 50.0%-80.0% of moisture loss expected from real FS dried on USDBs according to the literature (Koné and Strauss, 2004; Pescod, 1971). The dewatering performance of USDBs is known to be affected by the initial total solids and volatile solids content of FS and the applied loads (Koné and Strauss, 2004). Pescod (1971) observed drainage from real FS deposited onto USDBs with a sand and gravel filter of between 50.0% and 85.0% for sludge of initial total solids contents between 1.70% and 6.50% applied at a loading depth of 20.0cm. Given that the simulant sludge used in the drying test had a total solids content of approximately 45.57% - an increase of approximately 600% in comparison to the highest total solids of real FS recorded by Pescod (1971) – then it stands to reason that the drainage of moisture from the sludge would be significantly less. This is in keeping with Pescod's (1971) observation that sludge with a high initial total solids content took longer to drain resulting in evaporation becoming more significant. Therefore, more tests are required in which the total solids content of the simulant sludge is varied to assess how it dewaters in a covered, USDB and whether it is representative of the observed behaviour of real fS.

As the column drying test was conducted in conjunction with the ten day drying test with addition of UV light, similar fluctuation in temperature and relative humidity was observed, as shown in Figure 6-9.

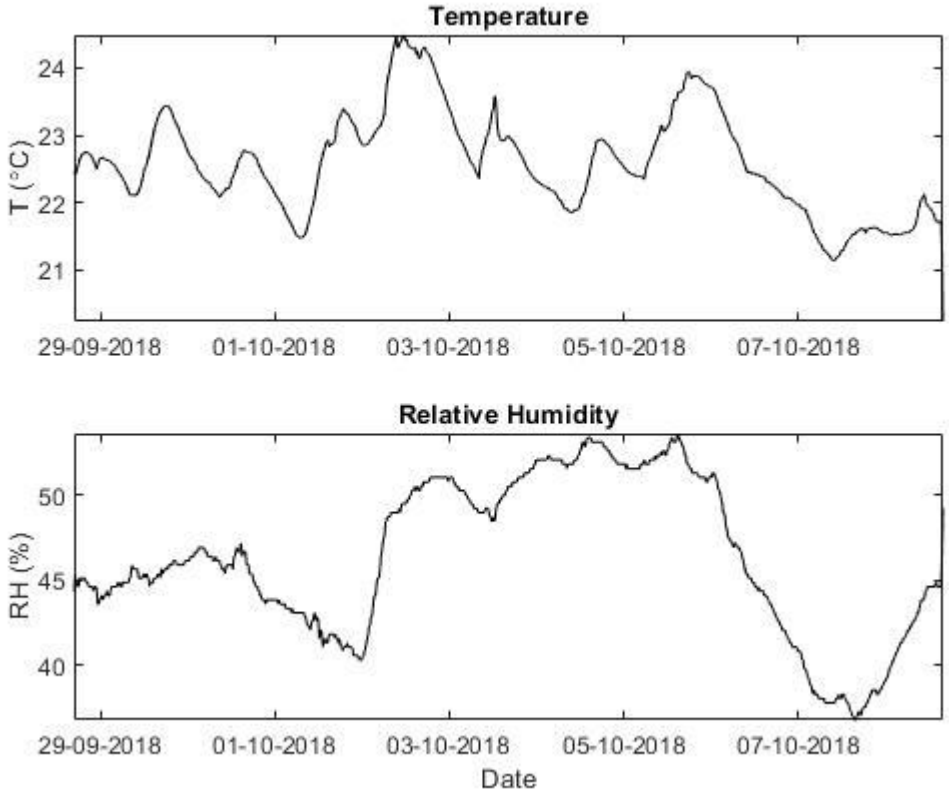
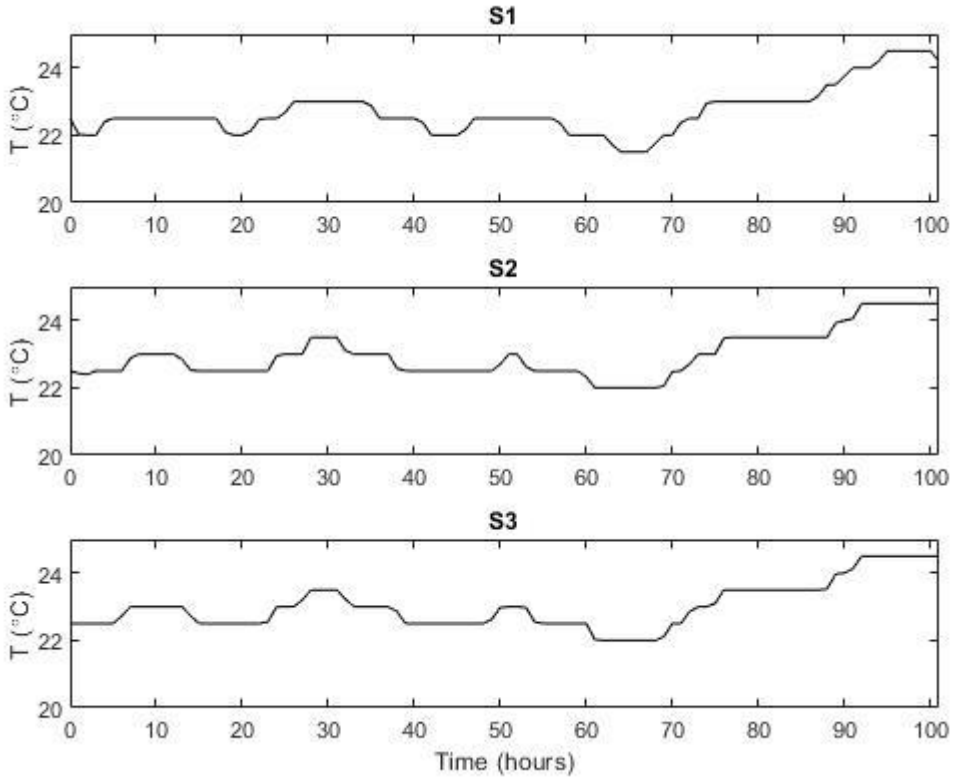
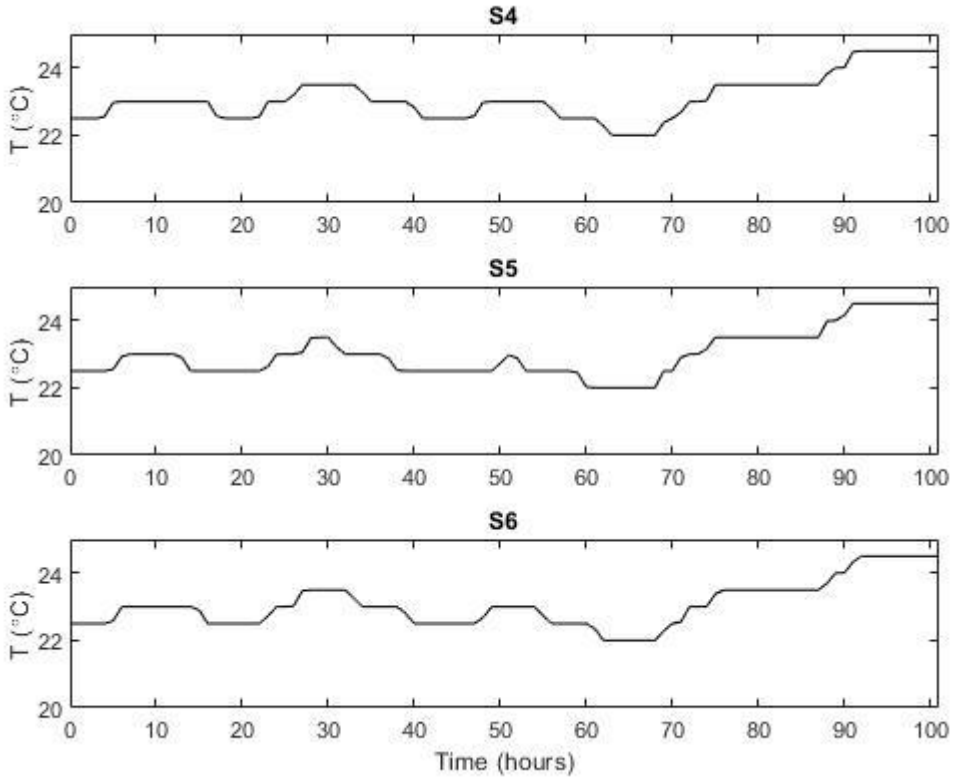


Figure 6-9. Temperature and relative humidity data for ten day small-scale column drying test in concrete UV lightbox

Due to a programming error the Thermochron 1-Wire DS1921G-F5 iButtons recorded temperature data in the bottom layer of simulant ND1 only for the first 100 hours (approximately four and a half days) of the ten day drying test. The hourly average temperature recorded by each iButtons is shown in Figure 6-10. Similar temperature trends within the sludge were recorded, fluctuating between approximately 21.5°C and 24.5°C. As may be observed in Figure 6-10, the temperatures recorded in the sludge were steady for long periods of time.



(a)



(b)

Figure 6-10. Average hourly temperature of bottom layer of faecal sludge simulant ND1 during first 100 hours of small-scale column drying test as recorded using Thermochron 1-Wire DS1921G-F5 iButtons where S is sensor

The temperature within the sludge was comparable to the ambient temperature of the concrete UV lightbox and appeared to follow a similar trend as shown in Figure 6-11. The iButton sensor data show that the sludge temperature lagged slightly behind the ambient temperature. A similar observation was made during the BH pilot test as it was noted that the real FS had "a slower response time to outside temperatures" (Way, 2013a). For the majority of the time that the iButton sensors logged data, the peak and lowest temperatures within the sludge appeared to exceed the peak and lowest ambient temperatures, as shown in Figure 6-11. However, after approximately 60 hours the difference in peak and lowest sludge temperature and peak and lowest ambient temperature was reduced. The peak temperature (24.5°C) recorded in the simulant sludge was less than that recorded in real FS (55.0°C) during the BH pilot study, likely due to the lower ambient temperature during the experiment. As observed during the BH pilot study, ambient air temperature appeared to be a reasonable proxy for sludge temperature during the small-scale column drying test. However, no conclusion could be drawn as to whether the heating of the sludge was isothermal, as it was not possible to monitor the surface temperature of the sludge.

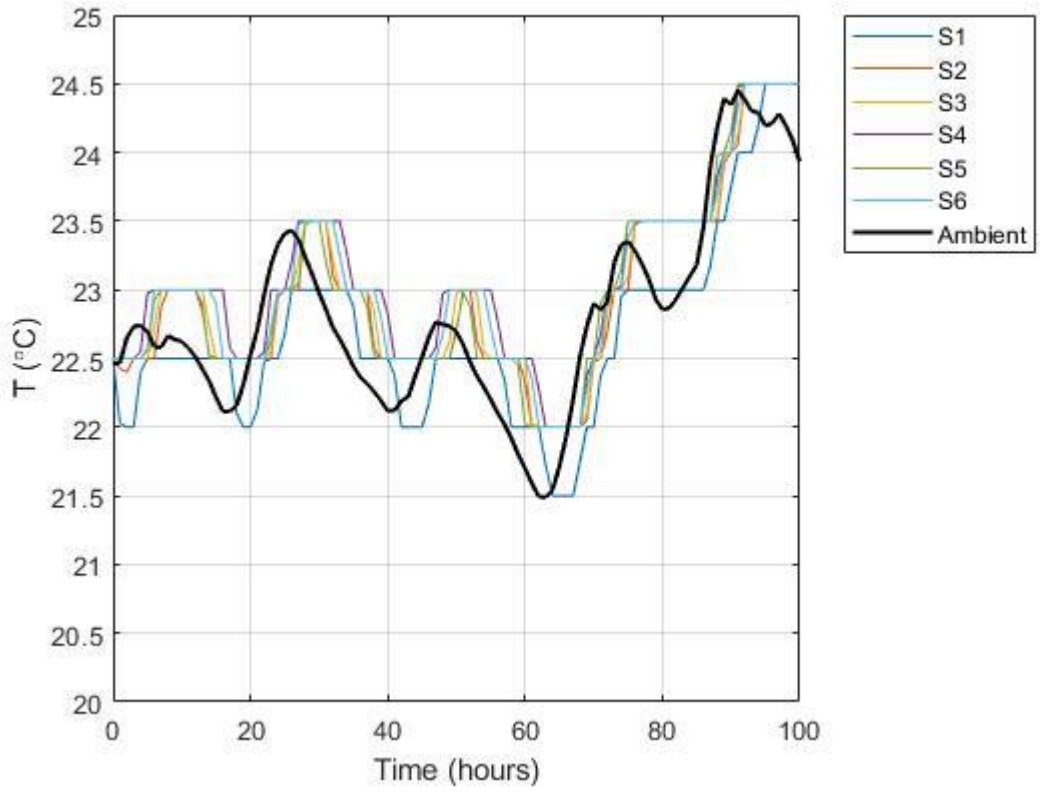


Figure 6-11. Comparison of average ambient hourly temperature in concrete UV lightbox and average hourly temperature of bottom layer of faecal sludge simulant ND1 during first 100 hours of small-scale column drying test where S is sensor

6.3.1 Drying Curves

Drying curves were plotted for the simulant sludge accounting for the loss of moisture by drainage. A curve of the evolution of average moisture content per hour was plotted as shown in Figure 6-12. The moisture-time plot was found to be relatively smooth with some noise towards the end of the drying test. The noise was likely due to the small changes in moisture content in the final stages of the drying test. The moisture-time plot was smoothed using a cubic smoothing spline in Matlab as recommended by Kemp *et al* (2001). The smoothing parameter p = 0.9889 was found to provide an adequate fit to the raw data without over or under smoothing, as shown in Figure 6-12.

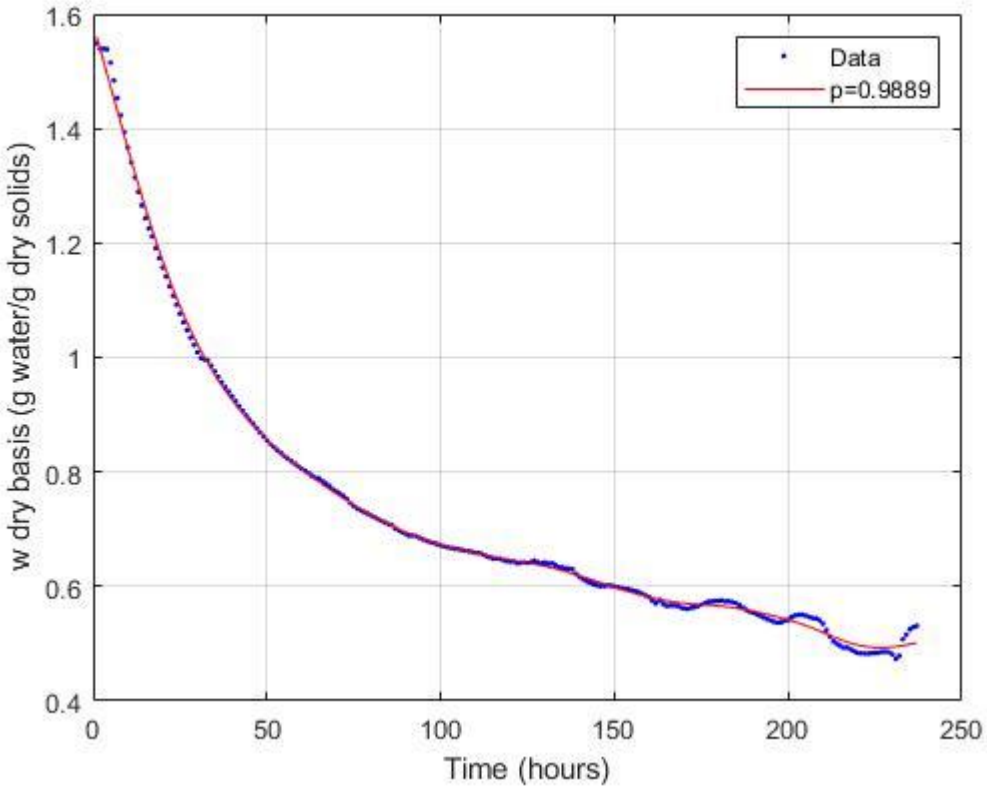


Figure 6-12. Evolution of moisture content with time for faecal sludge simulant ND1 where p is the best performing smoothing parameter for the cubic smoothing spline applied

A drying rate-time plot was obtained by differentiating the moisture content with respect to time as described by Equation 6-1. As shown in Figure 6-13, the drying rate curve exhibited a great deal of random noise, particularly towards the end of the drying test. According to Kemp *et al* (2004) this is "because the very low drying rates involved give small changes in recorded weight and moisture content, which are of the same order of magnitude as the noise from vibration of the weighing device." The smoothed moisture content-time curve was differentiated and fit to the drying rate curve; however, this resulted in an underestimation of the initial drying rate as shown in Figure 6-13. However, the smoothing for the latter part

of the curve (from t = 25 hours) was found to be reasonable. Further attempts to smooth the drying rate curve using a cubic smoothing spline, a robust local weighted regression (rloess) and Savitzky-Golay (sgolay) filtering resulted in under and over smoothing as shown in Figure 6-14.

From the raw data shown in Figure 6-13, no constant rate period could be identified and drying appeared to take place only in the falling rate period. As mentioned previously, denser, pasty sludges tend to dry predominantly in the falling rate period (Reyes et al., 2004). Given the low initial moisture content, simulant ND1 used for the drying column test was noted to have been thicker than previous batches of the same. This is likely the reason for the lack of constant rate period. Another potential cause could be the loss of free moisture through drainage, thus limiting the volume of free water available to dry by evaporation rom the surface of the simulant sludge. A further potential explanation for the lack of constant rate period could have been due to the formation of a crust on the surface of the sludge (Vaxelaire et al., 2000).

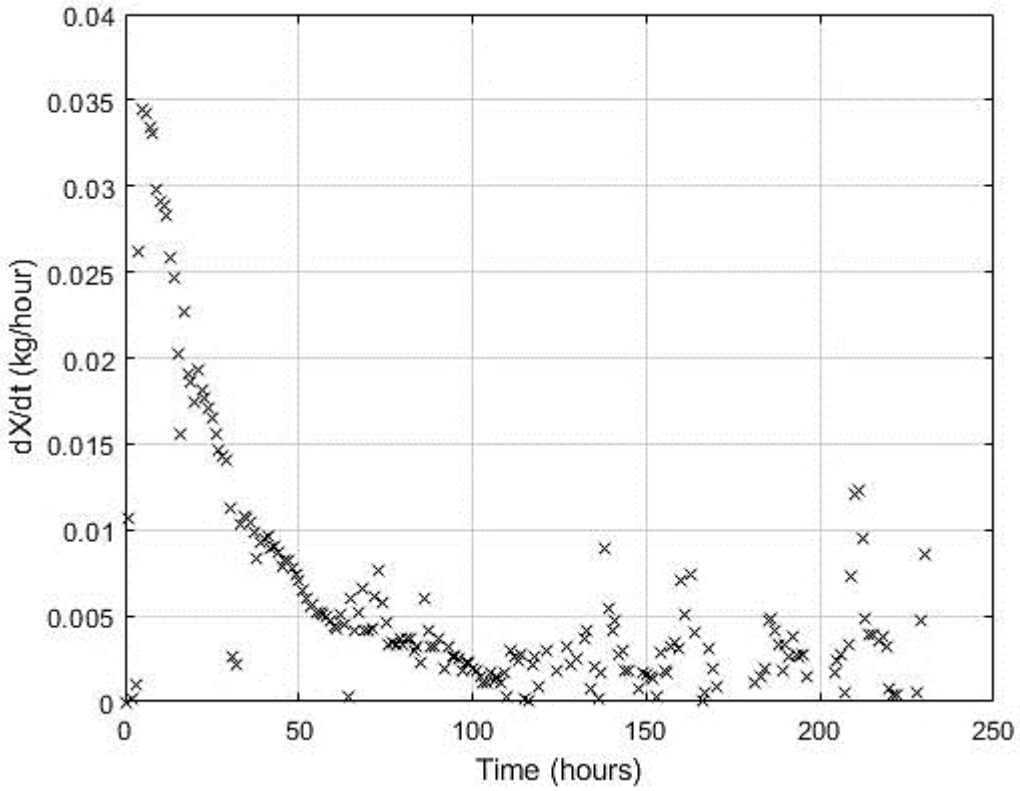


Figure 6-13. Evolution of drying rate with time for faecal sludge simulant ND1 where X_{cs}/dt is the differential of the moisture content-time curve smoothed with a cubic spline and smoothing parameter p = 0.000111

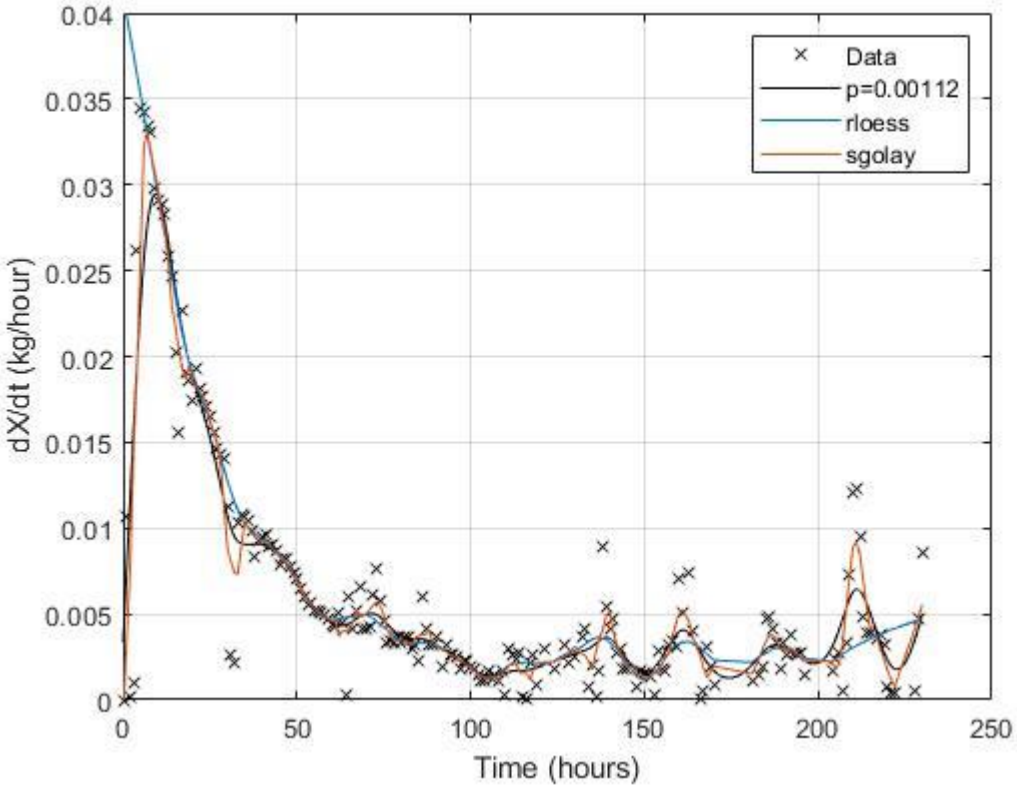


Figure 6-14. Smoothing of drying rate-time plot using cubic smoothing spline with smoothing parameter p = 0.00112, robust local weighted regression (rloess) with a span of 0.1 and Savitzky-Golay (sgolay) filtering with a span of 5.0 and polynomial degree of 1

The Kirscher plot of the experimental data shown in Figure 6-15exhibited similar scatter as that of the drying rate-time plot, particularly at the tail end of the plot. This was expected given the low drying rates and changes in moisture contents as explained previously. As noted by Kemp *et al* (2001) sharp transitions between the constant rate and falling rate periods of drying rarely exist in practice; therefore, it could be argued a constant rate period was present as indicated by the horizontal line at a drying rate of approximately 0.034kg hour^{-1} as shown on Figure 6-15. Assuming this to be the case, the critical moisture content may be identified as approximately 1.3kg water kg dry solids^{-1} therefore implying the majority of the moisture within the simulant sludge could be classed as bound water. However, it may be equally argued that no constant rate period exists and all of the water within the sludge could be classed as bound water. It was not possible to draw a conclusion in this regard.

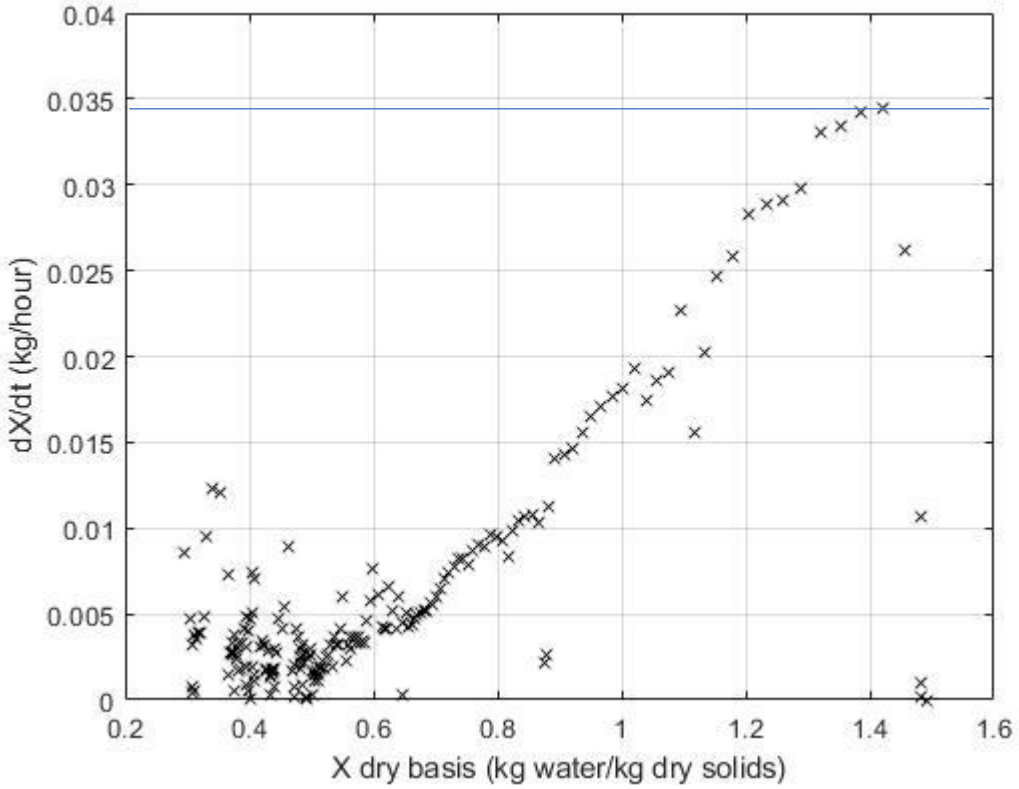


Figure 6-15. Kirscher plot for faecal sludge simulant ND1

6.4 Conclusions

Results of the preliminary drying tests in the climate chamber under constant temperature and relative humidity conditions of approximately 20°C and 60% were as expected. Simulants SS9 and BM1-4 all exhibited signs of bubble formation and expansion due to the presence of yeast. Simulants BM5, BM6, SE80 and ND1-4 did not exhibit any signs of bubbles. A thick, impermeable crust was observed by the end of the ten day drying period on the surface of simulants SS9 and BM1-6. Significant shrinkage appeared to occur in SE80 and ND4, perhaps due to the higher proportion of psyllium husk present in each recipe. Crusts were observed on SE80 and ND1-4 but they appeared to be less thick than those formed on simulants SS9 and BM1-6. Some small, micro-cracks were observed on the surfaces of simulants ND1-3. Based on characterisation results from the previous chapter and physical appearance, simulants ND1-4 were tested further with the addition of a radiation lamp in the climate chamber and simulant ND1 was used for a small-scale column drying test.

No conclusions could be drawn regarding the effect of radiation on the drying of simulant samples due to the proximity of the lamp to the samples as well as the consequent increase in temperature and decrease in relative humidity. No constant rate period could be identified for any of the simulant sludges tested. However, it was not possible to determine definitively that the simulant sludges dried only in the falling rate period. A lack of data recorded for

mass loss during the ten day drying test or crust formation may have been the cause of no identifiable constant rate period. Two falling rate periods were identified for each simulant tested. Simulants ND1-3 exhibited similar drying rates whilst the slowest drying simulant for all samples was simulant ND4.Tthis may have been due to consistency as simulant ND4 was more paste-like in comparison to simulants ND1-3. It may also have been due to the greater proportion of psyllium husk which would have likely increased the proportion of bound water within the sludge. All simulants were observed to shrink and develop crusts. Some micro cracks were observed on the surface of simulants ND1-3 by the end of the ten day drying test; however, simulant ND1 dried with a completely smooth surface.

A number of thin layer drying models were fit to the results for simulants ND1-4 dried with UV radiation in the climate chamber. The best performing models were the two term model and the model developed by Cai *et al* (2015). The models were applied over the full length of the mass ratio-time curves as it was assumed the simulants dried in the falling rate period only. Although the thin layer models applied were useful in smoothing the mass ratio-time curve, they are descriptive rather than predictive. It was not possible to apply a two stage model to any of the simulant sludges as a constant rate period could not be identified. Furthermore, no equilibrium moisture content was identifiable due to the humidity fluctuations.

Simulant ND1 was dried from liquidy to spadeable consistency in ten days in a small-scale column drying test, in keeping with the observations made about real FS dried on covered, USDBs. However, no runoff was observed over the duration of the ten day drying test which is not in keeping with the behaviour of real FS. However, the filter media was found to be damp at the end of the drying test, implying some moisture runoff from the sludge. The reduced runoff was likely due to the low initial moisture content of the simulant sludge which was the result of a calculation error in scaling up the simulant recipe. Thermocron iButton sensors logged temperature data in the bottom layer of the simulant sludge during the first 100 hours of the drying test. It was found there was some lag between the temperature of the sludge and the ambient temperature within the concrete UV lightbox, in keeping with observations of real FS dried in covered, USDBs. Furthermore, the ambient air temperature was found to be a reasonable indicator for sludge temperature, as concluded by Way (2013). Some shrinkage and crust formation, as well as some surface desiccation was observed following the removal of the simulant sludge at the end of the ten day drying test. No constant rate period could be conclusively identified from the drying curves for the ten day column drying test. This may have been due to the loss of any free moisture from the sludge through drainage. Alternatively the reduced initial moisture content may have resulted in a dense,

pasty sludge which is known to dry predominantly in the falling rate stage (Reyes et al., 2004).

The primary aim of the preliminary drying tests discussed in this chapter was to identify the presence of the three key behaviours exhibited by real FS when it is dried: shrinkage, cracking and crust formation. Shrinkage and crust formation were observed for all twelve FS simulants tested. Some micro-cracking was observed on the surfaces of simulants ND1-3, however, no significant cracking was observed in any simulant during any drying test. This may not be a severely limiting factor in the use of one of the developed simulants as there are instances in which real FS and wastewater sludge also do not exhibit cracking behaviour as discussed in Chapter 2. The best performing simulants in terms of both behaviour and representation of key properties of real type B FS were found to be simulants ND1-4.

7 Conclusions and Future Work

7.1 Summary of Thesis

FSM has become one of the major challenges in the sanitation sector globally. More efficient, effective and sustainable solutions are required across the entire FSM stream. The focus of this thesis was FS treatment by dewatering on USDBs. The aim was to develop a synthetic FS simulant to replicate the dewatering and drying properties of real FS dried on a covered USDB thus allowing safe, reproducible lab scale tests in order to improve understanding of FS dewatering and drying behaviour. This would enable the optimisation of the drying process and the development of more efficient technologies for sludge treatment by drying. Due to the limited available literature regarding the dewatering and drying behaviour of real FS, theory relating to the dewatering and drying of wastewater sludges was also reviewed. Common key material properties affecting dewatering and drying behaviour were highlighted and facilitated the selection of suitable FS simulants for characterisation tests. Twelve simulant sludges were developed and tested for the key properties found in the literature review. Drying tests were undertaken to identify simulants which exhibited key drying behaviour similar to that of real FS dried on covered, USDBs. Some thin layer drying models were successfully to results from the drying of FS simulants.

7.2 Thesis Conclusions

A number of research questions and objectives were presented at the beginning of this thesis. The extent to which these questions have been answered and the objectives met are discussed below.

RI. What are the key properties which affect the dewatering behaviour of FS?

OI. Provide a list of key properties which affect the dewatering behaviour of real FS deposited on USDBs.

This research question and objective were met in Chapter 2. Following an extensive literature review encompassing theory for wastewater sludge as well as some soils theory, a comprehensive list of properties which may affect the dewatering of FS treated on USDBs. Properties highlighted were the moisture distribution and moisture content; total solids concentration; organic matter concentration and composition; particle size distribution; ionic composition and concentration; degree of stabilisation; density and thermal properties. The extent to which some of these properties, for example thermal properties, influence the filterability and drying of FS was not addressed in the literature.

It was also observed that the majority of research into the drying of FS centred around either the impact of external conditions on the drying rate of FS or the impact that drying had on sludge properties. No literature was found regarding the impact of FS characteristics on the drying rate or drying behaviours commonly observed during FS treatment on USDBs. As research into the characteristics affecting the rate and extent of filterability of FS, this finding presents a useful avenue of further research. As the urban populations increase globally, land area availability for treatment options such as sludge drying beds is decreasing, and thermal drying or solar drying of sludge is becoming increasingly attractive for FS treatment. As such, understanding the FS properties which influence the rate of evaporation of moisture from sludge will be useful in increasing the efficiency of these new treatment options.

R2. Is it possible to replicate these properties is a synthetic simulant?

O2. Develop a number of simulants to represent the key properties highlighted in the literature review.

O3. Compare the key properties of the developed FS simulants with results for the same properties of real FS to determine the most representative simulant.

This research question and the objectives were partially met in this research in Chapters 3 and 4. In total eleven new simulants were developed at the University of Bath based on original simulants developed by PRG (2014) and Penn *et al* (2018). Simulants PRG SS9 and BM1-6 were adaptations of the original PRG (2014) simulant. Simulants ND1-4 were new simulants developed based on a list of ingredients which was assembled based on the literature review and the proportions of PRG (2014) and Penn *et al* (2018) simulants.

The four ND simulants were found to be more representative of the moisture content, total carbon, pH, density and dry and wet specific heat capacity for real type B FS than the PRG (2014) and Penn *et al* (2018) simulants. However, the total solids and organics contents were still found to be significantly higher than that reported for real type B FS. Consequently, the degree of stabilisation, as estimated by the volatile solids to total solids ratio remained significantly higher than for real type B FS.

One of the primary limitations in the development of FS simulants is a lack of characterisation data for real FS. As research into FSM continues it is anticipated that this limitation will be overcome and should enable better, more representative simulants to be developed. Furthermore, as understanding improved regarding the specific properties affecting the dewatering of FS by filtration and evaporation, the easier it should be to develop a suitable simulant. That said, based on the experience of this research, the likelihood of a universal

simulant to replicate all of the key properties affecting sludge dewatering behaviour is unlikely.

R3. Will a synthetic simulant replicate the drying behaviour of real FS deposited onto USDBs?

O4. Perform small-scale drying tests under controlled conditions to assess the drying behaviour of the FS simulants in comparison with that expected for real FS dried on an unplanted drying bed.

This research question and object were met in Chapters 5 and 6. All of the developed FS simulants were subject to a drying test in which their behaviour was observed. Crust formation was observed on simulants PRG SS9, BM1-4, SE80 and ND1-4 but not on simulants BM5 or BM6. Shrinkage was observed for all simulants however no cracking was observed which is not in keeping with the behaviour of real FS dewatered on USDBs. Bubble formation was significant in simulants PRG SS9 and BM1-4 due to the presence of yeast rendering them largely unsuitable for prolonged drying tests on drying beds.

As the best performing simulants in terms of key characteristics, the ND simulants were subject to a further drying test with the addition of a UV lamp to assess the impact of radiation on crust formation; however, results proved inconclusive as discussed in Section 7.3. No constant rate period was observed for any of the simulants as would be expected based on the literature detailed in Chapter 2, Section 2.2.

Simulant ND1 was also used for a small-scale column drying experiment to assess whether it could replicate the dewatering and drying expected for real FS deposited on a USDB. Simulant ND1 was found to dry from liquidy to spadeable consistency in ten days, in keeping with observations made in the field (Way 2013). That is to say the extent of dewatering appeared to have been similar to real FS; however, the rate of dewatering was significantly slower. According to the literature 50% - 80% of the volume of real FS deposited onto USDBs emerges as drained liquid within the first 24hrs – 48hrs (Way, 2013a; Pescod, 1971; Koné and Strauss, 2004). No runoff was observed from the sludge in the drying column throughout the duration of the drying test. Based on the final moisture content of the filter media, approximately of moisture was lost from the sludge which is significantly less than would be expected of real FS. This may have been caused by the low initial moisture content and high organics content of the simulant sludge, which may have limited the free water available to drain from the simulant sludge (Rawls et al., 1991; Rawls et al., 2003). Once again, no constant rate period was observed, and it was assumed that the dominant drying mechanism was evaporation based on the limited drainage.

R4. Can a model be developed to predict the necessary sludge retention time to achieve a particular moisture content under particular climate conditions?

O5. Perform small-scale drying bed tests to assess the rate of filtration and evaporation of synthetic simulant under different climate conditions.

This research and question were not met but steps were taken to achieve them as detailed in Chapter 6. Thin layer drying models were applied to the ND simulants dried with the addition of the UV lamp. The two-term thin layer drying model, Midilli-Kucuk thin layer drying model and Cai *et al* (2015) thin layer drying model were found to adequately describe the drying kinetics of simulants ND1-4 dried in a drying chamber with the addition of UV radiation. The drying rate of the simulant sludges was affected by their composition, in keeping with observations of the drying of real FS. Simulant ND4 was dense, and pasty and exhibited the slowest drying rate, in keeping with observations of real sludge (Reyes et al., 2004).

As no other tests were completed under other drying conditions and as such it was not possible to adapt the thin layer drying models. It was also intended to apply the two-stage drying models detailed in Chapter 2, Section 2.4; however, this was not achieved as none of the simulants dried exhibited a constant rate stage.

7.3 Contribution to Research

To the best of the author's knowledge, this is the first thesis to provide a comprehensive list of the properties which are likely to affect the drying of FS by filtration and evaporation. This will assist in future research regarding simulant development to replicate the key properties affecting the dewatering of sludge. A significant research gap was highlighted during the literature review in terms of the lack of research regarding the impact of sludge properties on its drying. This research also provides a framework for the development of FS simulants as well as an extensive literature review regarding existing FS simulants.

Perhaps the most important contribution to research is based on the experience of attempting to develop a FS simulant to replicate a wide range of properties. Although such a simulant could have significant impacts for the research of better, more efficient ways to treat waste, it may be concluded that focus should instead be on increasing the characterisation data for real FS and developing quick and cost effective methods for determining the properties of influent sludge to design efficient and sustainable treatment.

7.4 Recommendation for Further Work

The recommendations for further work are based directly on the limitations which presented themselves over the course of this work:

- As mentioned previously one of the major limitations in the development of a fully representative FS is the current lack of characterisation data for real FS. As such it is recommended that future work can focus on the development of sludge characterisation data bases and quick and cost-effective ways to determine the characteristics of influent sludge.
- It was not possible to draw conclusions for the representativeness of the simulant sludges regarding a number of properties – namely, thermal conductivity and particle size distribution, due to the limited sample size. Therefore, in order to continue the development of the ND simulants it is recommended that further testing is undertaken to develop a comprehensive database of the characteristics of the simulants.
- Several limitations in terms of the interpretation of results from the drying tests were also encountered. The temperature and relative humidity in the chamber were not offset to account for the increase in heat as a result of the addition of the radiation lamp. It was also observed during testing that the relative humidity was significantly reduced. The average temperature recorded was 22.5°C which is not a great deal higher than the set condition of 20.0°C. However, the average relative humidity of 46.0% was found to be much lower than the set condition of 60%. Furthermore, the relative humidity was noted to fluctuate a great deal. The addition of the UV lamp increased the temperature within the chamber but the moisture in the air entering the chamber remained the same, this is likely to have caused a decrease in relative humidity (Hall, 1970). The fluctuations observed were likely caused by the periodic turning off and on of the UV lamp in order to remove, photograph and weigh the synthetic FS samples. A further limitation was caused due to the inclusion of the lamp was its proximity to the samples.

  Therefore, it is recommended that further tests on the drying of simulant sludges employ a continuous weighing method rather than intermittent weighing. Doing so should provide enough data to identify the presence of any constant and falling rate periods ensuring an accurate picture of drying behaviour with various temperature and humidity conditions. It is also recommended that future drying tests do not include the use of a radiation lamp if drying in a climate chamber. Due to time and budgetary constraints it was not possible to develop a suitable model to predict the

drying the simulants. Therefore, further work could encompass the development of such a model. Due to the highly complex and non-homogenous nature of real FS, a two-stage model such as that developed by Efremov (1998) could prove useful in predicting drying time. An adapted version of the Penman equation for FS could also be developed. Future work should also encompass similar characterisation and drying tests for real FS in order to allow for comparison and ensure greater representativeness of real sludge.

- Based on the experience of this work, the author considers the primary limitation to the success of this work, aside from the attempt to replicate the properties of a highly complex and variable material, was the starting point. It is recommended that, should the reader wish to develop a simulant representative of real FS, they should begin with elemental analysis and build from that point.

References

Abu-Hamdeh, N.H. and Reeder, R.C., 2000. Soil thermal conductivity: effects of density, moisture content, salt concentration and organic matter. *Soil Science Society of America Journal*, 64(4), pp.1285–1290.

ACS, 2020a. *Climatic Chamber and Thermostatic Chamber* [Online]. Available from: https://acs.angelantoni.com/en/resources/climatic-chamber [Accessed 10 December 2020].

ACS, 2020b. *What is a test chamber and how does it work?* [Online]. Available from: https://acs.angelantoni.com/en/resources/what-is-a-test-chamber-and-how-does-it-work [Accessed 10 December 2020].

Ademiluyi, F.T. and Abowei, M.F.N., 2013. Theoretical model for predicting moisture ratio during spherical particles in a rotary dryer. *Modelling and Simulation in Engineering*, 2013, pp.1–7.

Afolabi, O.O.D., Sohail, M. and Wheatley, A.D., 2014. Hydrothermal treatment of human biowastes as an alternative sanitation strategy. *37th WEDC International Conference Sustainable Water and Sanitation Services for All in a Fast Changing World.* 15-19 September 2014, Hanoi. Loughborough, UK: WEDC.

Al-Atawneh, N., Mahmoud, N., van der Steen, P., and Lens, P.N.L., 2016. Characterisation of septage in partially sealed cesspit. *Journal of Water, Sanitation and Hygiene for Development*, 6(4), pp.631–639.

Alam, M.L., Saheed, S.M., Shinagawa, A. and Miyauchi, N., 1993. Chemical Properties of General Soil Types of Bangladesh. *Memoirs of the Faculty of Agriculture - Kagoshima University*, 29, pp.75–87.

Alamgir, M., 2010. *Present status of sewage sludge management in Khulna city.* Khulna, Bangladesh: NABOLOK and WaterAid Bangladesh.

de Albuquerque, C., 2010. *Human rights obligations related to access to safe drinking water and sanitation.* New York, NY: UN.

Alrhmoun, M., 2015. *Hospital wastewaters treatment: upgrading water systems plans and impact on purifying biomass.* Thesis (PhD). Universite de Limoges. Limoges, France.

Andriesse, J.P., 1988. Nature and management of tropical peat soils. *FAO Soils Bulletin*, 59.

Annan, K.A., 2000. *'We the Peoples': The role of the United Nations in the 21st century.* New York, NY: UN.

Arbogast, B. and Jaehyang, S., 2013. *Why faecal sludge management is serious business* [Online]. Available from: http://blogs.worldbank.org/water/why-fecal-sludge-management-serious-business [Accessed 1 November 2015].

ASTM, 2011. *E1269-11 Test method for determining specific heat capacity by differential scanning calorimetry*. West Conshohocken, PA: ASTM International.

ASTM, 2014. *D5334 Test method for the determination of thermal conductivity of soil and soft rock by needle probe procedure*. West Conshohocken, PA: ASTM International.

Badji, K., Dodane, P.H., Mbéguéré, M. and D, K., 2011. Traitement des boues de vidange: éléments affectant la performance des lits de séchage non plantés en taille réelle et les mécanismes de séchage. In: M. Mbéguéré, D. PH and D. Koné, eds. *Actes du symposium international sur la Gestion des Boues de Vidange* 30 juin-1 juillet 2009. Dakar, Sénégal: Eawag/Sandec, pp.32–38.

Bakare, B.F., 2014. *Scientific and management support for ventilated improved pit latrines (VIP) sludge content*. Thesis (PhD). University of KwaZulu-Natal. Durban, South Africa.

Bangladesh Bureau of Statistics (BBS) and UNICEF Bangladesh, 2015. *Progotir Pathey Bangladesh, Multiple Indicator Cluster Survey 2012-2013, Final Report*. Dhaka, Bangladesh: BBS/ UNICEF Bangladesh.

Bangladesh Bureau of Statistics (BBS) and UNICEF Bangladesh, 2019. *Progotir Pathey Bangladesh, Multiple Indicator Cluster Survey 2019, Survey Findings Report.* Dhaka, Bangladesh: BBS/ UNICEF Bangladesh.

Bassan, M., Mbeguere, M., Tchonda, T., Zabsonre, F., and Strande, L., 2013. Delivering Water, Sanitation and Hygiene Services in an Uncertain Environment - Characterization of Faecal Sludge During Dry and Rainy Seasons in Ouagadougou, Burkina Faso. *36th WEDC International Conference Delivering Water, Sanitation and Hygiene Services in an Uncertain Environment*, 1-5 July 2013, Nakuru. Loughborough, UK: WEDC, pp.216–221.

Bassan, M., Nguyen, V.A., Holliger, C. and Strande, L., 2014. Optimization of the Sampling and Laboratory Methods for Faecal Sludge. *Sandec News*, 15, pp.6–7.

Basu, P., 2010. Biomass Characteristics. *Biomass Gasification and Pyrolysis.* Oxford, England: Elsevier Inc, pp.27–63.

Basunia, M.A. and Abe, T., 1999. Moisture desorption isotherms of rough rice. *Journal of Food Engineering*, 42, pp.235–242.

van Bavel, C.H.M. and Hillel, D.I., 1976. Calculating potential and actual evaporation from a bare soil surface by simulation of concurrent flow of water and heat. *Agricultural Meteorology*, 17(6), pp.453–476.

Becker, B.R., Misra, A. and Fricke, B.A., 2013. Development of correlations for soil thermal conductivity. *Journal of Chemical Information and Modelling*, 53(9), pp.1689–1699.

Benali, M and Kudra, T., 2002. Thermal dewatering of diluted organic suspensions: process mechanism and drying kinetics. *Drying Technology*, 20(4–5), pp.935–951.

Berkman, E.T., 2018. Why Is Behaviour Change So Hard? [Online]. Available from: https://www.psychologytoday.com/intl/blog/the-motivated-brain/201803/why-is-behavior-change-so-hard [Accessed 14 May 2020].

Bhattacharjee, L., Saha, S.K. and Nandi, B.K., 2007. *Food-based nutrition strategies in Bangladesh*. Bangkok, Thailand: FAO.

Bill and Melinda Gates Foundation, 2019. *Reinvent the toilet challenge and expo: Strategy overview*. [Online]. Available from: https://www.gatesfoundation.org/what-we-do/global-growth-and-opportunity/water-sanitation-and-hygiene/reinvent-the-toilet-challenge-and-expo [Accessed 14 June 2018].

Blackett, I., Hawkins, P. and Heymans, C., 2014. *The missing link in sanitation service delivery - A review of fecal sludge management in 12 cities*. World Bank/WSP.

Boesch, A. and Schertenleib, R., 1985. *Emptying on-site excreta disposal systems: Field tests with mechanised equipment in Gaborone (Botswana).* Dübendorf, Switzerland: Eawag/Sandec.

Bockris, J.O., Hitchens, G.D. and Kaba, L., 1988. *Electrochemical processing of solid waste*. Houston, TX: NASA.

Boguslaw, U. and Lukasz, U., 2003. Thermal conductivity of soils - comparison of experimental results and estimation methods. *Water*, 20(4), p.10.

Boot, N., 2007. *Ecological Sanitation: A Concept*. Rugby, UK: Practical Action.

Bruce, D.M., 1985. Exposed-layer barley drying: Three models fitted to new data up to 150°C. *Journal of Agricultural Engineering Research*, 32, pp.337–341.

BSI, 1990. *BS 1377- 2 Methods of test for soils for civil engineering purposes - Part 2: Classification tests*. London, UK: BSI.

BSI, 2010. *BS ISO 13320 Particle size analysis - Laser diffraction methods (ISO13320)*. London, UK: BSI.

BSI, 2016a. *BS EN ISO 11357-1 Plastics - Differential scanning calorimetry (DSC) Part 1: General principles*. London, UK: BSI.

BSI, 2016b. *BS EN ISO 17892 Geotechnical investigation and testing — Laboratory testing of soil Part 4: Determination of particle size distribution*. London, UK: BSI.

BSI, 2018. *BS EN ISO 22007-1 Plastics - Determination of thermal conductivity and thermal diffusivity Part 1: General principles*. London, UK: BSI.

Buckley, C.A., Foxon, K.M., and Cottingham, R., 2012. *Data and design – mineralisation of sanitation wastes from community ablution blocks*. Durban, South Africa: UKZN.

Buckley, C.A., Foxon, K.M, Brouckaert, C.J., Rodda, N., Nwaneri, C., Balboni, E., Couderc, A. and Magagna, D., 2008. Scientific support for the design and operation of ventilated improved pit latrines (VIPs) and the efficacy of pit latrine additives. (Report NO TT 357/08). Pretoria, South Africa: WRC.

Byrne, A., 2012. *Sanitation in Developing Countries – Emptying Pit Latrines*. Thesis (MSc.). Trinity College Dublin, Dublin, Ireland.

Byrne, A., Sindall, L., Wang, L., de los Reyes III, F.L., and Buckley, C., 2017. What happens inside a pour-flush pit? Insight from comprehensive characterisation. *40th WEDC International Conference Local Action with International Cooperation to Improve and Sustain Water, Sanitation and Hygiene Services*. 24-28 July 2017, Loughborough. Loughborough, UK: WEDC.

Changara, M.C., Bangira, C., Sanyika, W.T., and Misi, S.N., 2018. Characterisation of pit latrine sludge from shackleton, a peri-urban residential area of Zimbabwe. *Journal of Water Sanitation and Hygiene for Development*, 8(3), pp.568-577.

C-Therm Technologies, 2020. Modified transient plane source (MTPS) technique [Online]. Available from: https://ctherm.com/products/tci_thermal_conductivity/how_the_tci_works/mtps/ [Accessed 23 January 2020].

Cai, Z.L., Ma, X.Q., Qing, X. and Yu, Z.S., 2015. Drying Kinetics and Characteristics of Sewage Sludge/Rice Straw Mixture. *Drying Technology*, 33(12), pp.1500–1509.

Cairns-Smith, S., Hill, H. and Nazarenko, E., 2014. *Urban sanitation: Why a portfolio of solutions is needed*. Boston, MA: BCG.

Carrington, E.G., 2001. *Evaluation of sludge treatments for pathogen reduction*. (CO 5026/1). Marlow, UK: European Commission Directorate-General Environment

Celma, A.R., Cuadros, F. and López-rodríguez, F., 2012. Convective drying characteristics of sludge from treatment plants in tomato processing industries. *Food and Bioproducts Processing*, 90(2), pp.224–234.

Chen, G., Lock Yue, P. and Mujumdar, A.S., 2002. Sludge Dewatering and Drying. *Drying Technology*, 20(4–5), pp.883–916.

Chen, G.W., Lin, W.W. and Lee, D.J., 1996. Capillary suction time (CST) as a measure of sludge dewaterability. *Water Science and Technology*, 34(3–4), pp.443–448.

Christensen, M.L., Keiding, K., Halkj, P.H. and Jorgensen, M.K., 2015. Dewatering in biological wastewater treatment: A review. *Water Research,* 82, pp.14–24.

CIAT and World Bank, 2017. *Climate-Smart Agriculture in Bangladesh*. Washington, DC: CIAT/World Bank.

Cofie, O. and Adamtey, N., 2009. Nutrient recovery from human excreta for urban and peri-urban agriculture. *34th WEDC International Conference Water, Sanitation and Hygiene: Sustainable Development and Multisectoral Approaches*, 18-22 May 2009, Addis Ababa. Loughborough, UK: WEDC.

Cofie, O., Kone, D., Rothenberger, S., Moser, D., and Zubruegg, C., 2009. Co-composting of faecal sludge and organic solid waste for agriculture: Process dynamics. *Water Research*, 43(18), pp.4665–4675.

Cofie, O., Van Rooijen, D., and Nikiema, J., 2014. Challenges and opportunities for recycling excreta for peri-urban agriculture in urbanising countries. In: B. Maheshwari, R. Purohit, H. Malano, V. P. Singh, & P. Amerasinghe, eds. *The security of water, food, energy and liveability of cities: challenges and opportunities for peri-urban futures*. Dordrecht, Netherlands: Springer, pp.301–310.

Cofie, O., Agbottah, S., Strauss, M., Esseku, H., Montangero, A., Awuah, E. and Kone, D., 2006. Solid-liquid separation of faecal sludge using drying beds in Ghana: Implications for nutrient recycling in urban agriculture. *Water Research*, 40(1), pp.75–82.

Coker, A.O., Sridhar, M.K.C., and Martins, E.A., 2003. Management of septic sludge in southwest Nigeria. *29th WEDC International Conference Towards the Millennium Development Goals*, 22-26 September 2003, Abuja. Loughborough, UK: WEDC.

Colón, J., Forbis-Stokes, A.A. and Deshusses, M.A., 2015. Anaerobic digestion of undiluted simulant human excreta for sanitation and energy recovery in less-developed countries. *Energy for Sustainable Development*, 29, pp.57–64.

Couderc, M.A., *Improving the rate of degradation of solids from VIP latrines*. Thesis (MSc.) Université Paul Cézanne, Marseille, France.

Danso-Boateng, E., Holdich, R.G., Shama, G., Wheatley, A.D., Sohail, M. and Martin, S.J., 2013. Kinetics of faecal biomass hydrothermal carbonisation for hydrochar production. *Applied Energy*, 111, pp.351–357.

Das, D., Keinath, T.M., Parker, D.S., Wahlberg, E.J., Das, D., Keinath, T.M., Parker, D.S. and Wahlberg, E.J., 1993. Floe breakup in activated sludge plants. *Water Environment Research*, 65(2), pp.138–145.

Deering, N., Briggs, K., Way, C. and Bryant, L., 2018. Characterisation of faecal sludge simulants for drying experiments. *41st WEDC International Conference Transformation Towards Sustainable and Resilient WASH Services*, 9-13 July, Nakuru. Loughborough, UK: WEDC.

Delicato, D.M.S., 1996. *Physical-chemical properties and sorption characteristics of peat*. Thesis (PhD). Dublin City University, Dublin, Ireland.

Dick, R.I. and Vesilind, P.A., 1969. The sludge volume index: what is it? *Journal (Water Pollution Control Federation)*, 41(7), pp.1285–1291.

Dickens, C., 2008. *A Tale of Two Cities*. Oxford, UK: Oxford University Press.

Dissanayaka, S.H., Hamamoto, S., Komatsu, T. and Kawamoto, K., 2013. Thermal properties for peaty soil under variable saturation and their correlation to mass transport parameters in gaseous and aqueous phases. *Research Report of Department of Civil and Environmental Engineering*, Saitama University, 39, pp.21–32.

Dodane, P.H. and Ronteltap, M., 2014. Unplanted drying beds. In: L. Strande, M. Ronteltap and D. Brdjanovic, eds. *Faecal Sludge Management*. London: IWA Publishing, pp.141–154.

Dong, Y., McCartney, J.S. and Lu, N., 2015. Critical Review of Thermal Conductivity Models for Unsaturated Soils. *Geotechnical and Geological Engineering*, 33(2), pp.207–221.

Duong, T.H., Vu, T.M.T., and Nguyen, V.A., 2014. Co-treatment of organic fractions of urban waste for energy recovery: a case study from Hanoi City, Vietnam. *37th WEDC International Conference Sustainable Water and Sanitation Services for All in a Fast Changing World*, 15-19 September 2014, Hanoi, Vietnam. Loughborough, UK: WEDC.

Efremov, G.I., 1998. Mathematical modeling of processes: Kinetics of convective drying of fibre materials based on solution of a diffusion equation. *Fibre Chemistry*, 30(6), pp.417–422.

Efremov, G.I., 2000. Generalised kinetics of drying of fibre materials. *Fibre Chemistry*, 32(6), pp.431–436.

Efremov, G.I., 2013. Describing of Generalised drying kinetics with application of experimental design. *Technical Sciences*, 16(4), pp.309–322.

Efremov, G.I. and Kudra, T., 2004. Calculation of the effective diffusion coefficients by applying a quasi-stationary equation for drying kinetics. *Drying Technology*, 22(10), pp.2273–2279.

Espinoza, O.A., Bond, B.H. and Loferski, J.R., 2007. Equilibrium moisture content during storage, manufacturing, and shipping of Bolivian wood products. *Forest Products Journal*, 57(6), pp.81–87.

Falade, K.O. and Abbo, E.S., 2007. Air-drying and rehydration characteristics of date palm (Phoenix dactylifera L.) fruits. *Journal of Food Engineering*, 79, pp.724–730.

Fanyin-Martin, A., Tamakloe, W., Antwi, E., Ami, J., Awarikabey, E., Apatti, J., Mensah, M., and Chandran, K., 2017. Chemical characterization of faecal sludge in the Kumasi metropolis, Ghana. *Gates Open Research*, 1(0), p.12.

FAO, 1988. *Report 3 Land Resources Database: Volume 2 - Soil, land and hydrological database*. Rome, Italy: FAO

FAO, 2014a. *AQUASTAT Country Profile - Bangladesh*. Rome, Italy: FAO.

FAO, 2014b. *Thailand - Food and nutrition security profiles*. Rome, Italy: FAO.

Farouki, O.T., 1986. Thermal Properties of Soils. *Series on Rock and Soil Mechanics*, 11, p.136.

Feachem, R.G., Bradley, D.J., Garelick, H. and Mara, D.D., 1983. *Sanitation and disease: Health aspects of excreta management*. Washington, DC: John Wiley and Sons, Inc.

Ferrasse, J.H., Arlabosse, P. and Lecomte, D., 2002. Heat, momentum, and mass transfer measurements in indirect agitated sludge dryer. *Drying Technology*, 20(4–5), pp.749–769.

Fisher, J.W., Pisharody, S., Wignarajah, K., Lighty, J.S., Burton, B., Edeen, M. and Davis, K.A., 1998. Waste incineration for resource recovery in bioregenerative life support systems. *SAE Technical Paper Series 981758*. Danvers, MA.

Flaga, A., 2005. Sludge drying. In: E. Plaza, E. Levlin and B. Hultman eds. *Proceedings of a Polish-Swedish Seminar, Integration and optimisation of urban sanitation systems*. 25-28 October 2003, Wislaw, Poland, pp.73–82.

Foxon, K., Buckley, C., Brouckaert, C., Bakare, B., Still, D. and Salisbury, F., 2011. How fast do pits and septic tanks fill up? Implications for design and maintenance. *What happens when the pit is full? Development in on-site faecal sludge management (FSM).* Durban, South Africa: WIN-SA and WRC.

Foxon, K., Buckley, C., Brouckaert, C., Wood, K. and Bakare, B., 2011. What is going on inside pits and septic tanks? The science of sludge decomposition. *What happens when the pit is full? Development in on-site faecal sludge management (FSM).* Durban, South Africa: WIN-SA and WRC.

Fumagalli, F. and Freire, J.T., 2007. Analysis of the drying kinetics of Brachiaria brizantha (Hochst. Stapf) grass seeds at different drying modes. *Drying Technology*, 25(9), pp.1437–1444.

Gardening Data, 2019. *John Innes Composts* [Online]. Available from: https://www.gardeningdata.co.uk/soil/john_innes/john_innes.php [Accessed 11 September 2019].

Gold, M., Cunningham, M., Bleuler, M., Arnheiter, R., Schönborn, A., Niwagaba, C., and Strande, L., 2018. Operating parameters for three resource recovery options from slow-pyrolysis of faecal sludge. *Journal of Water Sanitation and Hygiene for Development*, 8(4), pp.707–717.

Gold, M., Harada, H., Kimwaga, R., Niwagaba, C., and Strande, L., 2016. Working Towards Improved Faecal Sludge Dewatering. *Environmental Technology*. 37(21). pp.2802-2814

Gold, M., Harada, H., Therrien, J.-D., Nishida, T., Cunningham, M., Semiyaga, S., Fujii, S., Dorea, C., Nguyen, A., and Strande, L., 2018. Cross-country analysis of faecal sludge dewatering. *Environmental Technology*, 39(23), pp.3077–3087.

Gray, N.F., 2014. Capillary suction time (CST). In: S. Tarleton, ed. *Progress in Filtration and Separation*. London, UK: Academic Press, pp.659–668.

Greenberg, A., Clesceri, L., Eaton, A. and Franson, M., eds, 1976. *Standard methods for the examination of water and wastewater*. 14th ed. Washington, DC: APHA.

Gruter, H., Matter, M., Oehlmann, K.H. and Hicks, M.D., 1990. Drying of sewage sludge - An important step in waste disposal. *Water Science and Technology*, 22(12), pp.57–63.

Ha, S.R., Lee, D.H. and E, L.S., 1986. A mathematical model for the anaerobic digestion process applied to a mixture of night soil and septic tank sludge. *Water Science and Technology*, 18, pp.239–248.

Hall, C., 1980. *Drying and storage of agricultural crops*. Westport, CT: The AVI Publishing Company.

Hall, C.W., 1987. The evolution and utilisation of mathematical models for drying. *Mathematical Modelling*, 8, pp.1–6.

Hall, D., 1970. *Handling and storage of food grains in tropical and subtropical areas*. Rome, Italy: FAO.

Hanson, J.L., Edil, T.B. and Yesiller, N., 2000. Thermal properties of high water content materials. In: T.B. Edil and P.J. Fox, eds. *Geotechnics of high water content materials, ASTM STP 1374*. West Conshohocken, PA: ASTM, pp.137–151.

Harada, H., Dong, N.T. and Matsui, S., 2008. A measure for provisional-and-urgent sanitary improvement in developing countries: septic-tank performance improvement. *Water, Science and Technology*. 58(6). pp.1305–1311.

Heinss, U. and Koottatep, T., 1998. *Use of reed beds for faecal sludge dewatering*. Duebendorf, Switzerland: Eawag/Sandec.

Heinss, U., Larmie, S.A. and Strauss, M., 1998. *Solids separation and pond systems for the treatment of faecal sludges in the tropics*. Dübendorf, Switzerland: Eawag/Sandec.

Heinss, U., Larmie, S.A., and Strauss, M., 1999. *Characteristics of Faecal Sludges and their Solids-Liquid Separation*. Dübendorf, Switzerland: Eawag/Sandec.

Henderson, S.M., 1974. Progress in developing the thin layer drying equation. *Transactions of the American Society of Agricultural Engineers*, 17(6), pp.1167–1168.

Hibbert, D.B. and Gooding, J., 2006. Describing Data: Means and Confidence Intervals. *Data Analysis for Chemistry: An Introductory Guide for Students and Laboratory Scientists*. New York, NY: Oxford University Press, pp.39–65.

Hillel, D., 1998. *Environmental soil physics*. San Diego, CA: Academic Press.

Hoaglin, D.C. and Iglewicz, B., 1987. Fine-tuning some resistant rules for outlier labelling. *Journal of the American Statistical Association*, 82(400), pp.1147–1149.

Hossain, M.A. and Bala, B.K., 2002. Thin-layer drying characteristics for green chilli. *Drying Technology*, 20(2), pp.489–505.

Hukesflux Thermal Sensors, 2003. *TP02 non-steady-state probe for thermal conductivity measurement*. Delft, Netherlands: Hukesflux Thermal Sensors.

Imamul Huq, S.M. and Manzurul Hoque, A.F.M., 2012. Land and Soil Resources Database for Grass-Root Agricultural Development in Bangladesh. *International Conference on Advancing the Science and Technology of Soil Information in Asia - Launch of the Global Soil Partnership's Asia Soil Science Network and Global Soil Map*, 8-11 February 2012, Nanjing, China. Rome, Italy: FAO, pp.8–11.

Imamul Huq, S.M. and Shoaib, J.U.M., 2013. *Major Soil Types. The Soils of Bangladesh*. Dordrecht, Netherlands: Springer Netherlands, pp.41–49.

Ingallinella, A.M., Sanguinetti, G., Koottatep, T., Montanger, A. and Strauss, M., 2002. The challenge of faecal sludge management in urban areas--strategies, regulations and treatment options. *Water Science and Technology*, 46(10), pp.285–94.

Inyang, U.E., Oboh, I.O. and Etuk, B.R., 2018. Kinetic models for drying techniques - food materials. *Advances in Chemical Engineering and Science*, 8, pp.27–48.

Islam, R., Saha, U.K., Al Mamum, A., de la Brosse, N. and Stevens, 2016. Implications of inappropriate containment on urban sanitation and environment: a study of Faridpur, Bangladesh. *39th WEDC International Conference Ensuring Availability and Sustainable Management of Water and Sanitation for All*, 11-15 July 2016, Kumasi, Ghana. Loughborough, UK: WEDC.

ISO, 1991. *ISO 8301 Thermal insulation: Determination of steady-state thermal resistance and related properties - Heat flow meter apparatus*. Geneva, Switzerland: ISO.

Jere, M., Chidavaenzi, M., Nhandara, C., and Bradley, M., 1998. The effect of non-pathogenic bacteria on latrine sludge. *24th WEDC International Conference Sanitation and Water for All*, 31 August-4 September 1998, Islamabad, Pakistan. Loughborough, UK: WEDC.

Jin, B., Wilén, B. and Lant, P., 2004. Impacts of morphological , physical and chemical properties of sludge flocs on dewaterability of activated sludge. *Chemical Engineering Journal*, 98, pp.115–126.

Kaba, L., Hitchens, G.D. and Bockris, J.O.M., 1990. Electrochemical Incineration of Wastes. *Journal of the Electrochemical Society*, 137(5), pp.1341–1345.

Kalulu, K., Thole, B., Mkandawire, T., and Kululanga, G., 2018. Predictive modelling for characterisation of organics in pit latrine sludge from unplanned settlements in cities of Malawi. *Journal of Ecological Engineering*, 19(3), pp.141–145.

Karamouz, M., Nazif, S. and Falahi, M., 2012. *Hydrology and Hydroclimatology - Principles and Applications*. Boca Raton, FL: CRC Press.

Karr, P.R. and Keinath, T.M., 1978. Influence of Particle Size on Sludge Dewaterability. *Water Pollution Control Federation*, 50(8), pp.1911–1930.

Katukiza, A.Y., Ronteltap, M., Niwagaba, C.B., Foppen, J.W.A., Kansiime, F., and Lens, P.N.L., 2012. Sustainable sanitation technology options for urban slums. *Biotechnology Advances*, 30(5), pp.964–978.

Kelly, M.P. and Barker, M., 2016. Why is changing health-related behaviour so difficult? *Public Health*, 136, pp.109–116.

Kemp, I.C., 2011. Drying models, myths and misconceptions. *Chemical Engineering and Technology*, 34(7), pp.1057–1066.

Kemp, I.C., Fyhr, B.C., Laurent, S., Roques, M.A., Groenewold, C.E., Tsotsas, E., Sereno, A.A., Bonazzi, C.B., Bimbenet, J.J. and Kind, M., 2001. Methods for processing experimental drying kinetics data. *Drying Technology*, 19(1), pp.15–34.

Kemp, I.C. and Oakley, D.E., 2002. Modelling of particulate drying in theory and practice. *Drying Technology*, 20(9), pp.1699–1750.

Kemp, I.C. and Sohet, Q., 2010. Scale-up, optimisation and control of industrial batch fluidised bed dryers using multilevel theoretical models. *Drying Technology*, 28(5), pp.710–722.

Kengne, E.S., Magloire, K.I., Letah, W.A., Akoa, A., Nguyen-Viet, H. and Strande, L., 2013. Treatment Technology for Leachate from Faecal Sludge Drying Beds. *Sandec News*, p.10.

Kengne, I.M., Dodane, P.H., Akoa, A. and Koné, D., 2011. Vertical-flow constructed wetlands as sustainable sanitation approach for faecal sludge dewatering in developing countries. *Desalination*, 248(1–3), pp.291–297.

Kersten, M.S., 1949. Thermal properties of soils. *Engineering Experiment Station Bulletins*. (Bulletin No. 28). Minneapolis, MN: University of Minnesota.

Khanam, M. and Ahmed, S., eds, 2015. *Bangladesh Soil*. Banglapedia: National Encyclopedia of Bangladesh [Online]. Available from: http://en.banglapedia.org/index.php?title=Bangladesh_Soil#:~:text=Calcareous

Dark Grey Floodplain soils Occur extensively on,gleans as well in the Ganges tidal floodplain. [Accessed 16 December 2020].

Klingel, F., Montangero, A., Koné, D. and Strauss, M., 2002. *Faecal sludge management in developing countries: A planning manual*. Dübendorf, Switzerland: Eawag/Sandec.

Koné, D., Cofie, O., Zurbrügg, C., Gallizzi, K., Moser, D., Drescher, S. and Strauss, M., 2007. Helminth eggs inactivation efficiency by faecal sludge dewatering and co-composting in tropical climates. *Water Research*, 41(19), pp.4397–4402.

Koné, D., Gallizzi, K., and Drescher, S., 2004. Efficiency of Helminth egg removal in dewatered faecal sludge by co-composting. *30th WEDC International Conference People-Centred Approaches to Water and Environmental Sanitation*, 25-29 October 2004, Vientiane, Lao PDR. Loughborough, UK: WEDC.

Koné, D. and Peter, S., eds, 2008. *Sandec Training Tool 1.0 - Module 5: Faecal sludge management (FSM).* Dübendorf, Switzerland: Eawag/Sandec.

Koné, D. and Strauss, M., 2004. Low-Cost options for treating faecal sludges in developing countries - Challenges and performance. *9th International IWA Specialist Group Conference on Wetlands System for Water Pollution Control and 6th International IWA Specialist Group Conference on Waste Stabilisation Ponds*, 27 September-1 October 2004, Avignon, France. Dübendorf, Switzerland: Eawag/Sandec.

Koottatep, T., Polprasert, C., Thi, N., Oanh, K. and Strauss, M., 2001. Sludges from on-site sanitation systems - low-cost treatment alternatives. *IWA Conference on Water and Wastewater Management for Developing Countries*, 29-31 October 2001, Kuala Lumpur, Malaysia. London, UK: IWA Publishing, pp.29–31.

Koottatep, T., Surinkul, N., Paochaiyangyuen, R., Suebsao, W., Sherpa, M., Liangwannaphorn, C. and Panuwatvanich, A., 2012. *Assessment of faecal sludge rheological properties*. Bangkok, Thailand: Asian Institute of Technology

Koottatep, T., Surinkul, N., Polprasert, C., Kamal, A.S.M., Koné, D., Montangero, A., Heinss, U., and Strauss, M., 2005. Treatment of septage in constructed wetlands in tropical climate: lessons learnt from seven years of operation. *Water Science and Technology*, 51(9), pp.119–126.

Kopp, J. and Dichtl, N., 2000. Prediction of full-scale dewatering results of sewage sludges by the physical water distribution. *Water Science and Technology*, 42(9), pp.141–149.

Kopp, J. and Dichtl, N., 2001a. Characterisation. In: L. Spinosa and P.A. Vesilind, eds. *Sludge into Biosolids - Processing, Disposal, Utilisation*. London, UK: IWA Publishing, pp.19–40.

Kopp, J. and Dichtl, N., 2001b. The influence of free water content on sewage sludge dewatering. *Water Science and Technology*, 44(10), pp.177–183.

Kucuk, H., Midilli, A., Kilic, A. and Dincer, I., 2014. A review on thin-layer drying-curve equations. *Drying Technology*, 32(7), pp.757–773.

Kuffour, A.R., 2010. *Improving faecal sludge dewatering efficiency of unplanted drying bed*. Thesis (PhD). Kwame Nkrumah University of Science and Technology, Ashanti, Ghana.

Kuffour, A.R., Awuah, E., Anyemedu, F.O.K., Strauss, M., Koné, D. and Cofie, O., 2009. Effect of using different particle sizes of sand as filter media for dewatering faecal sludge. *Desalination*, 248(1–3), pp.308–314.

Lee, D.J. and Hsu, Y.H., 1995. Measurement of bound water in sludges: A comparative study. *Water Environment Research*, 67(3), pp.310–317.

Lemus-Mondaca, R., Betoret, N., Vega-Galvez, A. and Lara-Aravena, E., 2009. Dehydration characteristics of papaya (carica pubenscens): Determination of equilibrium moisture content and diffusion coefficient. *Journal of Food Process Engineering*, 32(2009), pp.645–663.

Lewis, W.K., 1921. The rate of drying of solid materials. *Journal of Industrial and Engineering Chemistry*, 13(5), pp.427–432.

Li, J., Fraikin, L., Salmon, T., Plougonven, E., Toye, D. and Léonard, A., 2016. Convective drying behavior of sawdust-sludge mixtures in a fixed bed. *Drying Technology*, 34(4), pp.395–402.

Lin, C.-Y., Bian, F.-Y., and Chou, J., 1999. Anaerobic co-digestion of septage and landfill leachate. *Bioresource Technology*, 68(3), pp.275–282.

Makununika, B., 2016. *Thermal Drying of Faecal Sludge from VIP latrines and Characterisation of Dried Faecal Material.* Thesis (MSc.). University of KwaZulu-Natal, Durban, South Africa.

Marzooghi, S. and Dentel, S.K., 2014. Sludge Drying Through Hydrophobic Membranes. In: S. Ahuja, ed. *Water Reclamation and Sustainability*. San Diego, CA: Elsevier, pp.361–390.

Midilli, A., Kucuk, H. and Yapar, Z., 2002. A new model for single-layer drying. *Drying Technology*, 20(7), pp.1503–1513.

Midilli, A., Kucuk, H and Yapar, Z., 2002. A new model for single layer drying. *Drying Technology*, 20(7), pp.1503–1513.

Mikkelsen, L.H. and Keiding, K., 2002. Physico-chemical characteristics of full scale sewage sludges with implications to dewatering. *Water Research*, 36(10), pp.2451–2462.

Mikkelsen, L.H., Mascarenhas, T. and Nielsen, P.H., 2002. Key parameters in sludge dewatering: Testing for the shear sensitivity and EPS content. *Water Science and Technology*, 46(10), pp.105–114.

Miller, A., Espanani, R., Junker, A., Hendry, D., Wilkinson, N., Bollinger, D., Abelleira-Pereira, J.M., Deshusses, M.A., Inniss, E. and Jacoby, W., 2015. Supercritical water oxidation of a model fecal sludge without the use of a co-fuel. *Chemosphere*, 141, pp.189–196.

Montangero, A. and Strauss, M., 2004. Faecal sludge treatment. Eawag/Sandec.

Mort, S., 2018. Why It Is So Hard To Change - The Neuroscience Made Simple [Online]. Available from: https://drsoph.com/blog/2018/7/6/why-it-is-so-hard-to-change-the-neuroscience-madesimple [Accessed 14 May 2020].

Mundo-Molina, M. and Coss-Gomez, E. De, 2017. Physical-Mathematical Foundations of the Penman Equation from the Perspective of the First Law of Thermodynamics and Energy Balance Theory. *Journal of Water Resource and Protection*, 9(12), pp.1377–1390.

Nakagiri, A., Niwagaba, C.B., Nyenje, P.M., Kulabako, R.K., Tumuhairwe, J.B., and Kansiime, F., 2017. Assessing ambient and internal environmental conditions of pit latrines in urban slums of Kampala, Uganda: effect on performance. *Journal of Water Sanitation and Hygiene for Development*, 7(1), pp.92–101.

Nartey, G.E., 2013. *Faecal Sludge Reuse in Urban and Peri-urban Crop Production*. Thesis (MPhil.). University of Ghana, Accra, Ghana.

National Centre for Biotechnology Information, n.d. PubChem Database. *Oleic Acid* [Online]. Available from: https://pubchem.ncbi.nlm.nih.gov/compound/Oleic-acid [Accessed 21 September 2019a].

National Centre for Biotechnology Information, n.d. PubChem Database. *Tritriacontanoic Acid* [Online]. Available from: https://pubchem.ncbi.nlm.nih.gov/compound/181572 [Accessed 11 September 2019b].

Nguyen, T.P., Hankins, N.P. and Hilal, N., 2007. A comparative study of the flocculation behaviour and final properties of synthetic and activated sludge in wastewater treatment. *Desalination*, 204(1–3), pp.277–295.

Niang, S., Gueye, A., Seck, A., Sonko, M., Gning, J.B., and Strande, L., 2012. From waste to resource - resource on FS drying beds in Dakar, Senegal. *Faecal Sludge Management 2,* 29-31 October 2012, Durban, South Africa. SuSanA.

Nikiema, J., Cofie, O., and Impraim, R., 2014. Technological Options for Safe Resource Recovery from Fecal Sludge. *Resource Recovery and Reuse*, 2, p.52.

Niwagaba, C., Mbéguéré, M., and Strande, L., 2014. Faecal sludge quantification, characterisation and treatment objectives. In: L. Strande, M. Ronteltap and D. Brdjanovic, eds. *Faecal Sludge Management*. London, UK: IWA Publishing, pp.18–43.

Novak, J., 2007. Dewatering of sewage sludge. *Drying Technology*, 24(10), pp.1257–1262.

Nwaneri, C.F., 2009. *Physico-chemical characteristics and biodegradability of contents of ventilated improved pit latrines (VIPs) in eThekwini Municipality*. Thesis (MSc.). University of KwaZulu-Natal, Durban, South Africa.

Nyaanga, D.M., Kabok, P.A., Mbuba, J., Abich, S.O., Eppinga, R., and Irungu, J., 2018. Faecal matter-sawdust composite briquette and pellet fuels: production and characteristics. *41st WEDC International Conference Transformation Towards Sustainable and Resilient WASH Services*, 9-13 July 2018, Nakuru, Kenya. Loughborough, UK: WEDC.

Nzouebet, W.A.L., Kengne, I.M., and Rechenburg, A., 2015. Does depth and sanitation type affec the quality of faecal sludge in the tropics? The case of Yaoundé, Cameroon. *Open Water Journal*, 3(1), pp.3–5.

Onabanjo, T., Kolios, A.J., Patchigolla, K., Wagland, S.T., Fidalgo, B., Jurado, N., Hanak, D., Manovic, V., Parker, A., McAdam, E., Williams, L., Tyrrel, S. and Cartmell, E., 2016. An experimental investigation of the combustion performance of human faeces. *Fuel*, 184, pp.780–791.

Onabanjo, T., Patchigolla, K., Wagland, S.T., Fidalgo, B., Kolios, A., McAdam, E., Parker, A., Williams, L., Tyrrel, S. and Cartmell, E., 2016. Energy recovery from human faeces via gasification: A thermodynamic equilibrium modelling approach. *Energy Conversion and Management*, 118, pp.364–376.

Opel, A., Bashar, M.K. and Ahmed, M.F., 2012. Faecal sludge management in Bangladesh: an issue that needs urgent attention. *Asia Regional Sanitation and Hygiene Practitioners Workshop*, 31 January – 2 February 2012, Dhaka, Bangladesh.

Orr, J.C., 1978. Cotton linters. *Journal of the American Oil Chemists' Society*, 55(5), p.A394.

OSRAM, 2006. *More than just light: Solutions in ultraviolet light*. OSRAM GmbH.

Armstrong, P. R., Casada, M. E. and Lawrence, J. 2012. Development of Equilibrium Moisture Relationships for Storage Moisture Monitoring of Corn. *Applied Engineering in Agriculture*, 28(5), pp.677–683.

Page, G.E., 1949. *Factors influencing the maximum rates of air drying shelled corn in thin layers.* Thesis (MSc.). Purdue University, West Lafayette, IN.

Paneque, M., De la Rosa, J.M., Kern, J., Reza, M.T. and Knicker, H., 2017. Hydrothermal carbonisation and pyrolysis of sewage sludges: What happens to carbon and nitrogen? *Journal of Analytical and Applied Pyrolysis*, 128, pp.314–323.

Parkinson, J., Lüthi, C. and Walther, D., 2014. *Sanitation 21: A planning framework for improving city-wide sanitation services*. IWA, Eawag/Sandec, GIZ.

Partners in Development and Pollution Research Group (PRG), 2012. *Tackling the challenges of full pit latrines Volume 1: Understanding sludge accumulation in VIPs and strategies for emptying full pits*. (WRC K5/1745). Gezina, South Africa: WRC.

Peng, G., Ye, F. and Li, Y., 2011. Comparative investigation of parameters for determining the dewaterability of activated sludge. *Water Environment Research*, 83(7), pp.667–671.

Penman, H.L., 1948. Natural evaporation from open water, bare soil and grass. *Proceedings of the Royal Society of London, Series A. Mathematical and Physical Sciences*, 193(1032), pp.120–145.

Penn, R., Ward, B.J., Strande, L. and Maurer, M., 2018. Review of synthetic human faeces and faecal sludge for sanitation and wastewater research. *Water Research*, 132, pp.222–240.

Peralta, P. and Bangi, A.P., 2005. Sensible heating approach to controlling the equilibrium moisture content of wood. *Forest Products Journal*, 55(12), pp.17–20.

Pescod, M.., 1971. Sludge Handling and Disposal in Tropical Countries. *Water Pollution Control Federation*, 43, pp.555–570.

Podichetty, J.T., Waliul Islam, M., Van, D., Foutch, G.L. and Johannes, A.H., 2014. Viscous heating analysis of simulant feces by computational fluid dynamics and experimentation. *Journal of Water, Sanitation and Hygiene for Development*, 4(1), pp.62-71.

Practical Action Bangladesh, 2014. *Situation Analysis Report of Faecal Sludge Management in Faridpur Municipality, Bangladesh*. Dhaka, Bangladesh: Practical Action.

Pradeep, R., Sarani.S, and S, S., 2017. Characteristics of faecal sludge generated from onsite systems located in Devanahalli. *Faecal Sludge Management 4*, 19-23 February 2017, Chennai, India. SuSanA.

PRG, 2014a. *Faecal Sludge Testing Data*. Durban, South Africa. Unpublished.

PRG, 2014b. *Selection of synthetic sludge simulant for the Bill and Melinda Gates Foundation's Reinvent the Toilet Fair: India 2014*. Durban, South Africa. Unpublished.

Quazi, A.R., 2006. Study on the use of human excreta in Bangladesh. In: M. Snel and J. Smet, eds. *The Value of Environmental Sanitation - Case Studies*. Delft, The Netherlands: IRC International Water and Sanitation Centre, pp.62–78.

Radford, J., Underdown, C., Velkushanova, K., Byrne, A., Smith, D., Fenner, R., Pietrovito, J. and Whitesell, A., 2015. Faecal sludge simulants to aid the development of desludging technologies. *Journal of Water, Sanitation and Hygiene for Development*, 5(3), pp.456–464.

Radford, J.T., 2011. *Fluidisation of synthetic pit latrine sludge*. Thesis (Meng.). University of Cambridge, Cambridge, UK.

Radford, J.T. and Fenner, R.A., 2013. Characterisation and fluidisation of synthetic pit latrine sludge. *Journal of Water, Sanitation and Hygiene for Development*, 3(3), pp.375–382.

Radford, J.T. and Sugden, S., 2014. Measurement of faecal sludge in-situ shear strength and density. *Water SA*, 40(1), pp.183–187.

Rawle, A., n.d. Basic principles of particle size analysis. *Technical Paper*, 44.

Rawls, W.J., Gish, T.J. and Brakensiek, D.L., 1991. Estimating soil water retention from soil physical properties and characteristics. *Advances in Soil Science*, 16, pp.213–234.

Rawls, W.J., Pachepsky, Y.A. and Ritchie, J.C., 2003. Effect of soil organic carbon on soil water retention. *Geoderma*. 116(1-2), pp.61–76.

Raynaud, M., Vaxelaire, J., Olivier, J., Dieude-Fauvel, E. and Baudez, J.-C., 2012. Compression dewatering of municipal activated sludge: Effects of salt and pH. *Water Research*, 46, pp.4448–4456.

Reddy, M., 2017. *Standard Operating Procedures* [Online]. Available from: http://prg.ukzn.ac.za/laboratory-facilities/standard-operating-procedures.

Rees, J.A., Smith, I. and Watson, J., 2015. Concentrations. *Introduction to Pharmaceutical Calculations*. London, UK: Pharmaceutical Press, pp.45–64.

Reyes, A., Eckholt, M., Troncoso, F. and Efremov, G., 2004. Drying kinetics of sludge from a wastewater treatment plant. *Drying Technology*, 22(9), pp.2135–2150.

Riffaldi, R., Sartori, F. and Levi-Minzi, R., 1982. Humic substances in sewage sludges. *Environmental Pollution Series B, Chemical and Physical*, 3(2), pp.139–146.

Rose, C., 2015. *Developing a nutrient recovery process for recovering nutrients in anaerobic digestate in low income countries*. Thesis (PhD). Cranfield University, Cranfield, UK.

Rose, C., Parker, A., Jefferson, B., Cartmell, E., Parker, A., Jefferson, B. and Characterization, E.C.T., 2015. The characterisation of feaces and urine: A review of the literature to inform advanced treatment technology. *Critical Reviews in Environmental Science and Technology*, 45(17), pp.1827–1879.

Saeed, I.E., Sopian, K. and Abidin, Z.Z., 2008. Drying Characteristics of Roselle: Mathematical Modeling and Drying Experiments. Agricultural Engineering Internation: CIGR Journal, 10(1).

Salomone, L.A. and Kovacs, W.D., 1984. Thermal Resistivity of Soils. *Journal of Geotechnical Engineering*, 110(3), pp.375–389.

Salomone, L.A., Kovacs, W.D. and Kusuda, T., 1984. Thermal performance of fine-grained soils. *Journal of Geotechnical Engineering*, 110(3), pp.359–374.

Sanin, F.D. and Vesilind, P.A., 1999. A comparison of physical properties of synthetic sludge with activated sludge. *Water Environment and Research*, 71(2), pp.191–196.

Sawalha, O. and Scholz, M., 2010. Modeling the relationship between capillary suction time and specific resistance to filtration. *Journal of Environmental Engineering*, 136(9), pp.983–991.

Schmid, H., 2014. Measuring a small number of samples and the 3 v fallacy: shedding light on confidence and error intervals. *IEEE Solid-State Circuits Magazine* [Online], 6, pp.52–58. Available from: https://doi.org/10.1109/MSSC.2014.2313714.

Schoebitz, L., Bassan, M., Ferré, A., Vu, T.H.A., Nguyen, V.A., and Strande, L., 2014. FAQ: Faecal sludge quantification and characterization – field trial of methodology in Hanoi, Vietnam. *37th WEDC International Conference Sustainable Water and Sanitation Services for All in a Fast Changing World*, 15-19 September 2014, Hanoi, Vietnam. Loughborough, UK: WEDC.

Schoebitz, L., Bischoff, F., Ddiba, D., Okello, F., Nakazibwe, R., Niwagaba, C., Lohri, C., and Strande, L., 2016. *Results of faecal sludge analyses in Kampala, Uganda*. Dübendorf, Switzerland: Eawag/Sandec.

Schoebitz, L., Nguyen, V.A., Tran, H.H., Dang, T.H., and Strande, L., 2014. *From research to implementation: Component 1 - Waste supply and availability - Hanoi*. Dübendorf, Switzerland: Eawag/Sandec

Schoebitz, L., Niwagaba, C., Francis, O., Bischoff, F. and Strande, L., 2014. FAQ : Faecal sludge quantification and characterization – Kampala. *Sandec News*, 15, pp.12–13.

Schoebitz, L., Niwagaba, C., and Strande, L., 2014. *From research to implementation: Component 1 - Waste supply and availability report - Kampala*. Dübendorf, Switzerland: Eawag/Sandec.

Scholz, M., 2005. Review of recent trends in capillary suction time (CST) dewaterability testing research. *Industrial and Engineering Chemistry Research*, 44(22), pp.8157–8163.

Seck, A., Gold, M., Niang, S., Mbeguere, M., Diop, C., and Strande, L., 2015. Faecal sludge drying beds: increasing drying rates for fuel resource recovery in Sub-Saharan Africa. *Journal of Water, Sanitation and Hygiene for Development*, 5(1), pp.72–80.

Semiyaga, S., Okure, M.A.E., Niwagaba, C.B., Nyenje, P.M., and Kansiime, F., 2017. Dewaterability of faecal sludge and its implications on faecal sludge management in urban slums Faecal sludge pre-treatment by dewatering. *International Journal of Environmental Science and Technology*, 14(1), pp.151–164.

Septien, S., Mugauri, T., Singh, A. and Inambao, F., 2017. *Drying of faecal sludge using solar thermal energy*. Durban, South Africa: PRG.

Septien, S., Mugauri, T.R., Singh, A. and Inambao, F., 2018. Solar drying of faecal sludge from pit latrines in a bench-scale device. *41st WEDC International Conference*

Transformation Towards Sustainable and Resilient WASH Services, 9-13 July 2018, Nakuru, Kenya. Loughborough: WEDC.

Septien, S., Singh, A., Mirara, S.W., Teba, L., Velkushanova, K. and Buckley, C.A., 2018. 'LaDePa' process for the drying and pasteurization of faecal sludge from VIP latrines using infrared radiation. *South African Journal of Chemical Engineering*, 25, pp.147–158.

Serio, M.A., Cosgrove, J.E., Stapleton, T.J., Nalette, T.A., Ewert, M.K., Lee, J., Fisher, J., Hartford, E., Hartford, E., President, V., Energy, C., Materials, C., Hartford, E., Engineer, P.D., Technology, A., Systems, S., Locks, W., Locks, W., Support, L., Analyst, T.S., Branch, B. and Field, M., 2016. Torrefaction Processing for Human Solid Waste Management. *46th International Conference on Environmental Systems*. Vienna, Austria.

Shao, L., He, Peipei, Yu, G. and He, Pinjing, 2009. Effect of proteins, polysaccharides, and particle sizes on sludge dewaterability. *Journal of Environmental Sciences*, 21(1), pp.83–88.

Shisana, O., Labadarios, D., Rehle, T., Simbayi, L., Zuma, K., Dhansay, A., Reddy, P., Parker, W., Hoosain, E., Naidoo, P., Hongoro, C., Mchiza, Z., Steyn, N., Dwane, N., Makoae, M., Maluleke, T., Ramlagan, S., Zungu, N., Evans, M., Jacobs, L., Faber, M., 2013. *The South African national health and nutrition examination survey (SANHANES-1)*. Cape Town, South Africa: HSRC Press.

Sigma-Adrich, n.d. *Product information. Casein* [Online]. Available from: https://www.sigmaaldrich.com/content/dam/sigma-aldrich/docs/Sigma/Product_Information_Sheet/c3400pis.pdf [Accessed 9 December 2019].

Simpson, W., 1973. Predicting equilibrium moisture content of wood by mathematical models. *Wood and Fiber*, 5(1), pp.41–49.

Sindall, R.C., Byrne, A., and Buckley, C.A., 2017. Rheology of sludge in pour-flush toilets: understanding the requirements for pit emptying technology design. *40th WEDC International Conference Local Action with International Cooperation to Improve and Sustain Water, Sanitation and Hygiene* (Wash) Services, 24-28 July 2017, Loughborough, UK. Loughborough, UK: WEDC.

Skinner, S.J., Studer, L.J., Dixon, D.R., Hillis, P., Rees, C.A., Wall, R.C., Cavalida, R.G., Usher, S.P., Stickland, A.D. and Scales, P.J., 2015. Quantification of wastewater sludge dewatering. *Water Research*, 82, pp.2–13.

Smith, J.K. and Vesilind, P.A., 1995. Dilatometric measurement of bound water in wastewater sludge. *Water Research*. 29(12), pp.2621–2626.

Smollen, M., 1986. Dewaterability of municipal sludges - a comparative study of specific resistance to filtration and capillary suction time as dewaterability parameters. *Water SA*, 12(3), pp.127–132.

Smollen, M., 1990. Evaluation of municipal sludge drying and dewatering with respect to sludge volume reduction. *Water Science and Technology*, 22(12), pp.153–161.

Sobeck, D.C. and Higgins, M.J., 2002. Examination of three theories for mechanisms of cation-induced bioflocculation. *Water Research*, 36, pp.527–538.

Sonko, E.H.M., Mbéguéré, M., Diop, C., Niang, S., and Strande, L., 2014. Effect of hydraulic loading frequency on performance of planted drying beds for the treatment of faecal sludge. *Journal of Water, Sanitation and Hygiene for Development*, 4(4), p.633.

Spence, J., Buttsworth, D., McCabe, B.K., Baillie, C., Antille, D.L. and Carter, B., 2018. Investigation into thin layer drying rates and equilibrium moisture content of abattoir paunch waste. *Renewable Energy*, 124, pp.95–102.

Stevens, L., Islam, R., Morcrette, A., de la Brosse, N. and al Mamun, A., 2015. Faecal sludge management in Faridpur, Bangladesh: scaling up through service level agreements. *38th WEDC International Conference Water, Sanitation and Hygiene Services beyond 2015: Improving Access and Sustainability,* 26-30 July 2015, Loughborough, UK. Loughborough, UK: WEDC.

Strande, L., Ronteltap, M. and Brdjanovic, D., eds, 2014. *Faecal Sludge Management*. London, UK: IWA Publishing.

Strande, L., Schoebitz, L., Bischoff, F., Ddiba, D., Okello, F., Englund, M., Ward, B. and Niwagaba, C., 2018. Methods to reliably estimate faecal sludge quantities and qualities for the design of treatment technologies and management solutions. *Journal of Environmental Management*, 223, pp.898–907.

Strauss, M., Drescher, S., Zurbrügg, C., Montangero, A., Cofie, O., and Dreschel, P., 2003. *Co-composting of Faecal Sludge and Municipal Organic Waste*. Dübendorf, Switzerland: Eawag/Sandec.

Strauss, M. and Heinss, U., 1995. Faecal sludge treatment. *Sandec News*, 1, pp.2–13.

Strauss, M. and Heinss, U., 1997. *Workshop on Faecal Sludge Treatment,* 3-5 December 1997, Sogakope, Ghana. Dübendorf, Switzerland: Eawag/Sandec.

Strauss, M., Larmie, S.A. and Heinss, U., 1997. Treatment of sludges from onsite sanitation - low cost options. *Water Science and Technology*, 35(6), pp.129–136.

Strauss, M., Larmie, S.A., Heinss, U., and Montangero, A., 2000. Treating faecal sludges in ponds. *Water Science and Technology*, 42(10), pp.283–290.

Strauss, M. and Montangero, A., 2002. *FS management - review of practices, problems and initiatives*. Dübendorf, Switzerland: Eawag/Sandec.

SuSanA, 2017. *Definition of sanitation value chain* [Online]. Available from: https://forum.susana.org/166-definitions-glossaries-dictionaries-mapping-tools/21715-definition-of-sanitation-value-chain [Accessed 4 March 2019].

Svoboda, E., 2017. *Why Is It So Hard to Change People's Minds?* [Online]. Available from: https://greatergood.berkeley.edu/article/item/why_is_it_so_hard_to_change_peoples_minds [Accessed 14 May 2020].

Talla, A., Sezawo, R., and Ngohe-Ekam, P.-S., 2017. Characterization of Depotted Faecal Sludge into the Environment and Design of a Suitable Treatment System: Case of Nomayos Area in Yaounde City. *British Journal of Applied Science & Technology*, 21(2), pp.1–12.

Tamakloe, W., 2014. *Characterization of faecal sludge and analysis of its lipid content as a viable precursor for biodiesel production*. Thesis (BSc). Kwame Nkrumah University of Science and Technology, Kumasi, Ghana.

Tao, T., Peng, X.F. and Lee, D.J., 2005. Thermal drying of wastewater sludge: Change in drying area owing to volume shrinkage and crack development. *Drying Technology*, 23(3), pp.669–682.

Tayler, K., 2018. *Faecal sludge and septage treatment: A guide for low and middle income countries*. Rugby, UK: Practical Action.

Taylor, C., 2012. *An investigation into the potential of faecal sludge for plant production.* Thesis (MSc). University of KwaZulu-Natal, Durban, South Africa.

Teba, L., Septien, S., Velkushanova, K., Pocock, J., and Buckley, C., 2015. Rheology, extrusion and pelletisation of faecal sludge. *Faecal Sludge Management 3*, 19-21 January 2015, Hanoi, Vietnam. SuSanA.

Tilley, E., Ulrich, L., Lüthi, C., Reymond, P. and Zurbrügg, C., 2014. *Compendium of Sanitation Systems and Technologies*. 2nd ed. Dübendorf, Switzerland: Eawag/Sandec.

Todman, L.C., Eekert, van Eekert, M.H.A., Templeton, M.R., Hardy, M., Gibson, W.T., Torondel, B., Abdelahi, F., and Ensink, J.H.J., 2015. Modelling the fill rate of pit latrines in Ifakara, Tanzania. *Journal of Water, Sanitation and Hygiene for Development*. 5(1), pp.100–107.

Togrul, I.T. and Pehlivan, D., 2002. Mathematical modelling of solar drying of apricots in thin layers. *Journal of Food Engineering*, 55(3), pp.209–216.

Tsang, K.-W.R. and Vesilind, P.A., 1990. Moisture Distribution in Wastewater Sludges. *Water Science and Technology*, 22(12), pp.135–142.

Turtoi, M., 2013. Ultraviolet light potential for wastewater disinfection. Annals. Food Science and Technology, 14(1), pp.153–164.

UNDG, 2003. *Indicators for Monitoring the Millennium Development Goals: Definitions, Rationale, Concepts and Sources*. New York, NY: UNDG.

United Nations, 2015a. *The Millennium Development Goals Report*. New York, NY: UN.

United Nations, 2015b. *The United Nations World Water Development Report 2015: Water for a Sustainable World*. Paris, France: UNESCO.

United Nations, 2016. *The Millennium Development Goals Report* [Online]. Available from: http://www.undp.org/content/undp/en/home/sdgoverview/mdg_goals/ [Accessed 22 August 2016].

UNU-INWEH, 2015. *Fast Facts* [Online]. Available from: http://sanitationdrive2015.org/resources-2/fast-facts/ [Accessed 1 November 2015].

US EPA, 1980. *Septage Management*. Cincinnati, OH: US EPA.

US EPA, 1984. *Septage Treatment and Disposal*. Cincinnati, OH: US EPA.

US EPA, 1994. *Guide to Septage Treatment and Disposal*. Cincinnati, OH: US EPA.

Vaxelaire, J., Bongiovanni, J.M., Mousques, P. and Puiggali, J.R., 2000. Thermal drying of residual sludge. *Water Research*, 34(17), pp.4318–4323.

Vaxelaire, J. and Cezac, P., 2004. Moisture distribution in activated sludges: A review. *Water Research*, 38(9), pp.2214–2229.

Vaxelaire, J. and Puiggali, J.R., 2002. Analysis of the drying of residual sludge: From the experiment to the simulation of a belt dryer. *Drying Technology*, 20(4–5), pp.989–1008.

Velkushanova, K., 2015. Properties of faecal sludge from different on-site sanitation facilities in Durban, South Africa. *Faecal Sludge Management 3*, 19-21 January 2015, Hanoi, Vietnam. SuSanA.

Velkushanova, K., Tartibu, L., Rajan, M. and Buckley, C.A., 2017. Development and testing of faecal sludge simulants. *Faecal Sludge Management 4*, 19-23 February 2017, Chennai, India. SuSanA.

Velkushanova, K., Zuma, L., and Buckley, C., 2012. Mechanical Properties of Faecal Sludge. *Faecal Sludge Managment 2*, 29-31 October 2012, Durban, South Africa. SuSanA.

Vesilind, P.A., 1988. Capillary measure suction of time sludge as a fundamental dewaterability. *Water Pollution Control Federation*, 60(2), pp.215–220.

Vesilind, P.A., 1994. The role of water in sludge dewatering. *Water*, 66(1), pp.4–11.

Vesilind, P.A. and Hsu, C.-C., 1997. Limits of sludge dewaterability. *Water Science and Technology*, 36(11), pp.87–91.

Vincent, J., Forquet, N., Molle, P. and Wisniewski, C., 2012. Mechanical and hydraulic properties of sludge deposit on sludge drying reed beds (SDRBs): Influence of sludge characteristics and loading rates. *Bioresource Technology*, 116, pp.161–169.

Vincent, J., Molle, P., Wisniewski, C., and Liénard, A., 2011. Sludge drying reed beds for septage treatment: Towards design and operation recommendations. *Bioresource Technology*, 102(17), pp.8327–8330.

de Vries, D.A., 1963. Thermal properties of soil. In: W. van Wijk, ed. *Physics of Plant Environment*. New York: John Wiley and Sons, Inc., pp.210–235.

Wang, L.K., Shammas, N.K. and Hung, Y.-T., eds, 2007. *Biosolids treatment processes* [Online]. 6th ed. Totowa, NJ: Humana Press. Available from: https://doi.org/10.1017/CBO9781107415324.004.

Ward, B.J., Morgenroth, E. and Strande, L., 2017. Dewatering characterisation of synthetic faecal sludge project. *Sandec News*, 18, p.8.

Ward, B.J., Traber, J., Gueye, A., Morgenroth, E. and Strande, L., 2019. Evaluation of conceptual model and predictors of faecal sludge dewatering performance in Senegal and Tanzania. *Water Research*, 167, pp.1–13.

Ward, B.J., Yacob, T.W. and Montoya, L.D., 2014. Evaluation of solid fuel char briquettes from human waste. *Environmental Science and Technology*, 48(16), pp.9852–9858.

WaterAid, 2012. *Exploring an effective, doable and affordable sludge management solution in rural areas of Bangladesh: a proposal for action research*. London, UK: WaterAid.

WaterAid, 2018. Urban pit waste management. *WaterAid Technology Briefs*. London, UK: WaterAid.

Way, C., 2012. *Investigation into faecal sludge management (FSM)*. Bath, UK: BH Bath.

Way, C., 2013a. *FSM Action Research, Bangladesh - Final Project Report*. Bath, UK: BH Bath.

Way, C., 2013b. *FSM Bangladesh: Action Research Project*. Bath, UK: BH Bath.

White, G.M., Ross, I.J. and Poneleit, C.G., 1981. Fully-exposed drying of popcorn. *Transactions of the American Society of Agricultural Engineers*, 24, pp.466–468.

WHO and UNICEF JMP, 2014. *Progress on Drinking Water and Sanitation*. Geneva, Switzerland: WHO.

WHO and UNICEF JMP, 2016. *WASH in the 2030 Agenda*. Geneva, Switzerland: WHO.

WHO and UNICEF JMP, 2017a. *Estimates on the Use of Water, Sanitation and Hygiene in Bangladesh*. Geneva, Switzerland: WHO.

WHO and UNICEF JMP, 2017b. *Progress on Drinking Water, Sanitation and Hygiene: 2017 Update and SDG Baselines*. Geneva, Switzerland: WHO.

Wignarajah, K., Hogan, J.A., Fisher, J.W., Delzeit, L. and Fox, T.G., 2008. Evaluation of the microwave enhance freeze drying. *38th International Conference on Environmental Systems,* 29 June-2 July 2008, San Francisco, CA. Warrendale, PA: SAE International. Paper No. 2008-01-2051.

Wignarajah, K., Litwiller, E., Fisher, J. and Hogan, J., 2006. Simulated human feces for testing human waste processing in space systems. *36th International Conference on Environmental Systems,* 17-20 July 2008, Norfolk, VA. Warrendale, PA: SAE International. Paper No. 2006-01-2180.

Wilén, B.-M., 1995. *Effect of different parameters on settling properties of activated sludge.* (Rapport 1995:4). VA-teknik Chalmers.

Wilkinson, D., 2019. *Why some behaviours are really difficult to change and what to do about it: CEOS Theory* [Online]. Available from: https://www.oxford-review.com/why-some-behaviours-are-really-difficult-to-change-and-what-to-do-about-it/ [Accessed 1 December 2020].

Wood, K., 2013. *Transformation of faecal sludge in VIPs: Modelling fill rate with an unsteady-state mass balance*. Thesis (MSc). University of KwaZulu-Natal, Durban, South Africa.

World Bank, 2020a. *Bangladesh: Climate Data - Historical* [Online]. Available from: https://climateknowledgeportal.worldbank.org/country/bangladesh/climate-data-historical [Accessed 24 January 2020].

World Bank, 2020b. *South Africa: Climate Data - Historical* [Online]. Available from: https://climateknowledgeportal.worldbank.org/country/south-africa/climate-data-historical [Accessed 24 January 2020].

World Bank, 2020c. *Thailand: Climate Data - Historical* [Online]. Available from: https://climateknowledgeportal.worldbank.org/country/thailand/climate-data-historical [Accessed 24 January 2020].

WSSCC, 2000. Vision 21: A Shared Vision for Hygiene, Sanitation and Water Supply and a Framework for Action. *Proceedings of the 2nd World Water Forum*. The Hague, Netherlands: WSCC.

Xanthopoulos, G., Oikonomou, N. and Lambrinos, G., 2007. Applicability of a single-layer drying model to predict the drying rate of whole figs. *Journal of Food Engineering*, 81, pp.553–559.

Yaldýz, O. and Ertekýn, C., 2001. Thin layer solar drying of some vegetables. *Drying Technology*, 19(3–4), pp.583–597.

Yeo, R. and Welchel, D., 1994. *Synthetic fecal fluid compound*. United States Patent 5356626. October 18, 1994.

Young, C., 2019. *Excel solver - which solving method should I use?* [Online]. Available from: https://www.engineerexcel.com/excel-solver-solving-method-choose/ [Accessed 3 November 2019].

Yousuf, T.B. and Mahmud, W., 2011. *A study on situation analysis and business model development of faecal sludge management of Faridpur Municipality*. Faridpur, Bangladesh: Practical Action Bangladesh and WaterAid Bangladesh.

Zhao, P., Liu, C., Qu, W., He, Z., Gao, J., Jia, L., Ji, S. and Ruan, R., 2019. Effect of temperature and microwave power levels on microwave drying kinetics of zhaotong lignite. *Processes*, 7(2).

Ziebell, F., Gold, M., Matovu, J., Maiteki, J., Niwagaba, C., and Strande, L., *Dewatering of faecal sludge with geotextiles: Results from laboratory and bench-scale experiments in Kampala, Uganda*. Kampala, Uganda: Eawag/Sandec.

Zuma, L., 2013. *Development of a synthetic faecal sludge*. Dübendorf, Switzerland: Eawag/Sandec.

Zuma, L., Velkushanova, K., and Buckley, C., 2015. Chemical and thermal properties of dry VIP latrine sludge. *Water SA*, 41(4), pp.534–540.

Appendices

A Buro Happold Bath and WaterAid Bangladesh Faecal Sludge Treatment Project

Appendix A provides a detailed outline of the pilot study performed by Buro Happold (BH) Bath for WaterAid Bangladesh. The first section provides context for faecal sludge management (FSM) in Bangladesh. In section A.2 potential FS treatment options are outlined. In the final section details regarding the selection of covered, unplanted sludge drying beds (USDBs) for the treatment of FS and an outline of the findings of the pilot study are provided.

A.1 Faecal Sludge Management in Bangladesh

An estimated 69.1% of the Bangladesh population has access to an improved sanitation facility (including shared facilities) (WHO and UNICEF JMP, 2017a). However, resource and capacity gaps coupled with a lack of awareness regarding the risks of disposal of untreated sludge have resulted in poor FSM in the country (Opel et al., 2012). A study by Blackett *et al* (2014) on faecal waste flows in Dhaka revealed that the majority of on-site sanitation facilities were left to overflow or abandoned. The remaining facilities were emptied and the untreated sludge was disposed of into the environment (Blackett et al., 2014). A report into the status of sewage sludge[1] management in Khulna found that there was no suitable location for the safe disposal of sludge and instead it was often disposed untreated into drains or water bodies (Alamgir, 2010). Another small-scale study found that, while FS has been used in agriculture and for biogas production in Bangladesh for a number of years, it has often been undertaken in an unhygienic way causing serious health and environmental risks (Quazi, 2006). Hence, WaterAid Bangladesh approached BH Bath in 2012 to develop a FS treatment solution for rural and small town areas which was effective, practical, affordable and did not affect existing toilet technology in the country (Way 2013b).

The way in which the effectiveness of a treatment technology is determined can depend to a large degree on the intended end-use or disposal option. For example, if the intention is to use the treated sludge as a soil conditioner in agriculture then moisture content reduction and nutrients such as nitrogen are of key importance (Montangero and Strauss, 2004; Strande et al., 2014). Where the treated sludge is to be disposed of into the environment

[1] Sewage sludge in this case is taken to refer faecal sludge as defined in Strande *et al* (2014) as there is no official sewage system and the majority of households in Khulna rely on onsite sanitation systems to deal with their waste, thus the sludge is not transported through a sewer.

emphasis is placed on properties such as chemical oxygen demand (COD) or ammonium (NH_4). Pathogen reduction remains key for any treatment, but the level required will, again, depend on the intended end-use or disposal. 'Practical' is taken in this case to refer to the necessity for a selected treatment option to be context-specific and carefully selected based on available infrastructure as well as on human and financial resources in order to ensure the sustainability of the technology (United Nations, 2015b). Affordability of FS treatment options is of key importance from a user perspective and may significantly influence access, particularly for the poor (United Nations, 2015b). The willingness to pay for water, sanitation and hygiene (WASH) services is highly variable so it is more likely that greater affordability will lead to a greater willingness to pay for the service. Although affordability is highly context specific, it can be assumed that an affordable sanitation service is one which can be accessed without compromising "the ability to pay for other essential needs guaranteed by human rights such as food, housing and health care" (de Albuquerque, 2010).

That the selected treatment option should not affect existing toilet technology is relevant to a context-specific approach to the design of a FS treatment system. Often, plans for sanitation systems are "prepared with aspirational objectives, without a realistic consideration of what is actually achievable given the availability of existing resources and ignoring existing investments" (Parkinson et al., 2014). It seems relatively intuitive to avoid the introduction of new technology given that changing already entrenched behaviour is notoriously difficult (Mort, 2018; Svoboda, 2017; Berkman, 2018; Wilkinson, 2019; Kelly and Barker, 2016). This is provided that existing facilities remain functioning effectively and may be considered improved according to the new WHO and UNICEF JMP (2017) sanitation ladder. This context-specific approach is encompassed in the new emphasis within the SDGs on encouraging countries to account for national circumstances when planning sanitation-related goals for the SDG era (WHO and UNICEF JMP, 2016).

According to WaterAid Bangladesh (2012) there have been previous attempts to address FSM involving ecological sanitation (ecosan)[2]. However, although technically sound, it was found to be practically unfeasible given the soci0-economic[3] and cultural contexts of the country (WaterAid, 2012). These findings hold when the breakdown of improved sanitation facilities

[2] Ecosan operates on the principle that human excreta is not a waste product but useful for agriculture as it contains the necessary nutrients to act as a soil conditioner (from faeces) or fertiliser (from urine) (Boot, 2007). There are two types of ecosan toilet: dehydrating (urine diversion) and composting. Excreta is meant to undergo primary treatment through increase of pH or reduction of moisture content in the latrine and will either require removal for secondary treatment or will undergo secondary treatment within the facility as is the case for an alternating pit system, for example.

[3] Ecosan latrines involve a high investment cost – up to 7 times that of a 'normal' private latrine – and were considered unaffordable for the majority of the population (WaterAid, 2012; Way, 2013a).

is considered. According to the most recent survey by the Bangladesh Bureau of Statistics (BBS) and UNICEF Bangladesh (2019) 55.4% of those with access to improved facilities relied on latrines including VIP latrines, flush/pour flush latrines connected to pits and pit latrines with a slab. 7.2% reported having a sewer connection (mostly in urban areas) and 22.8% relied on septic tanks. Only 0.1% reported the use of a composting toilet (Bangladesh Bureau of Statistics (BBS) and UNICEF Bangladesh, 2019). Furthermore, this statistic has remained unchanged since the previous BBS and UNICEF Bangladesh survey for the year 2012/2013 (Bangladesh Bureau of Statistics (BBS) and UNICEF Bangladesh, 2015). To ensure that existing toilet technology was not affected the team at BH Bath elected to focus on (semi-)centralised treatment options[4] which are discussed in the following section.

A.2 (Semi-)Centralised Treatment Options for Faecal Sludge

(Semi-)centralised options for the treatment of FS were compiled from Tilley *et al* (2014) and Strande *et al* (2014) and are shown in Figure A-1. Boxes in bold are those that are relatively well-established for FS dewatering while italics is used to denote treatment technologies which are in development or for which pilot trials have been reported in the literature. Boxes made of dashed lines are used to indicate treatment technologies commonly used for wastewater treatment that have not yet been trialled for FS but may be applicable. Technologies are grouped according to their treatment objectives/goals: pre-treatment; solid-liquid separation; stabilisation; pathogen reduction; effluent; and end product/end-use. Where one technology can accomplish several objectives, it spans across the objectives, for example, co-composting achieves both stabilisation and pathogen reduction of dewatered sludge and therefore spans both these objectives in Figure A-1. Vertical flow and subsurface horizontal flow constructed wetlands can be used for solid-liquid separation by filtration, stabilisation and pathogen reduction as well as for effluent polishing following initial dewatering and therefore spans across most treatment objectives (Klingel et al., 2002).

[4] Semi-centralised treatment and centralised treatment refer to wastewater and FS treatment technologies that are generally appropriate for large user groups (i.e., neighbourhood to city level applications) (Tilley et al., 2014). They require either the distribution of fresh water to the consumer or the removal and transport of FS and wastewater from the consumer to the point of treatment.

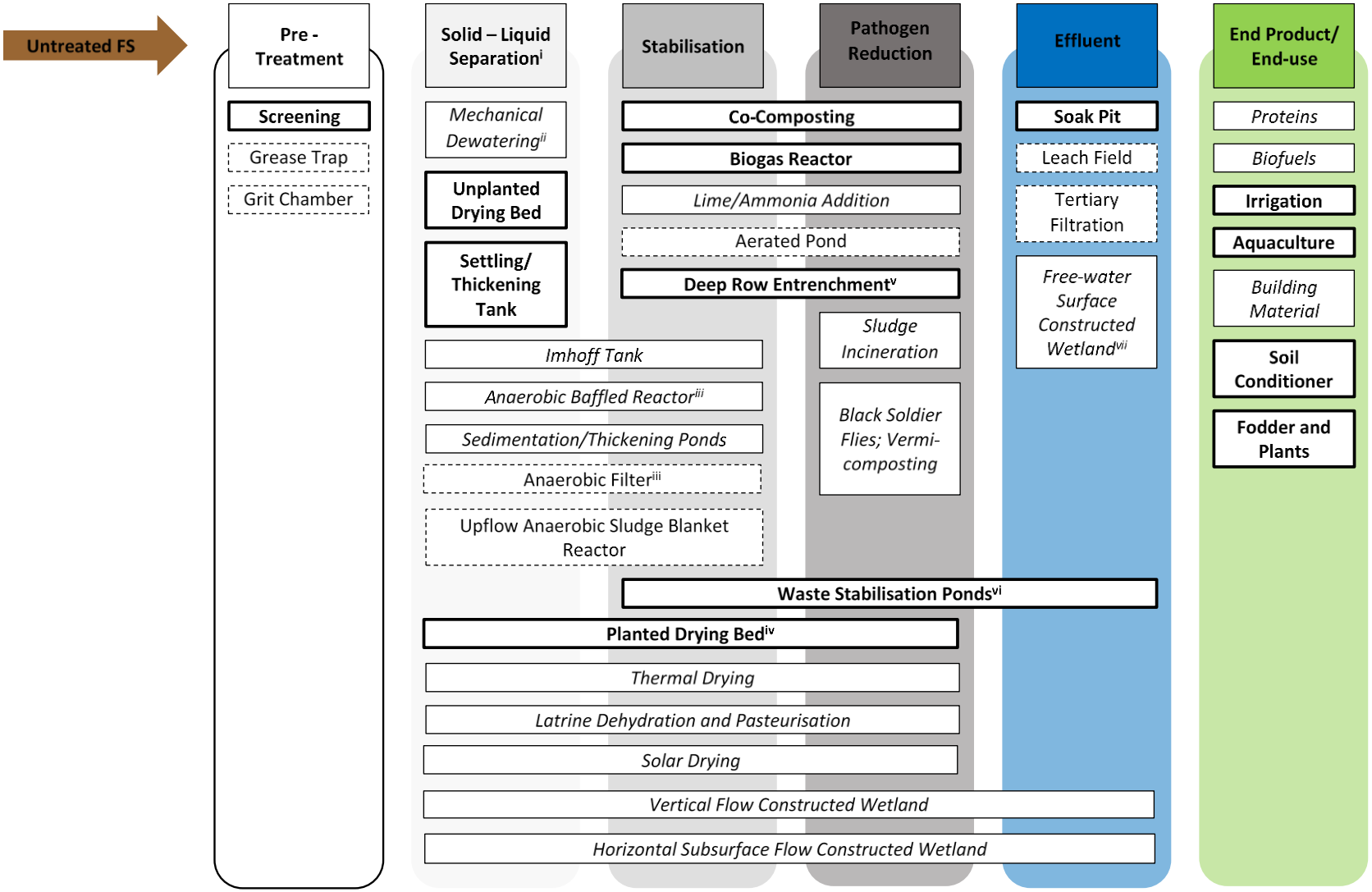


Figure A-1. (Semi-)centralised wastewater and faecal sludge treatment options where FS is faecal sludge

[a] Solid-liquid separation includes dewatering of FS by gravity separation (i.e. settling), filtration, evaporation and evapotranspiration, centrifugation and heat drying.
[ii] Mechanical dewatering technologies include the belt filter press, centrifuge, frame filter press and screw press.
[iii] Gravity separation may take place in an integrated settling compartment or it may take place in a separate settler or another preceding technology, for example, existing septic tanks (Tilley et al., 2014).
[iv] Planted drying beds may sometimes be referred to as vertical flow constructed wetlands in FS literature (Strande et al 2014). However, the two technologies operate differently. According to Tilley *et al* (2014) vertical flow constructed wetlands are designed for pre-settled wastewater (due to issues with clogging) while planted drying beds are designed specifically for FS which can be applied directly onto the previous layer. Additionally, the operation of each technology is different: vertical flow constructed wetlands are dosed intermittently with wastewater causing the filter to go through stages of being saturated and unsaturated leading to different phases of aerobic and anaerobic conditions (Tilley et al., 2014). In contrast, FS is deposited onto a planted drying bed all at once and ventilation pipes connected to the drainage system contribute to aerobic conditions in the filter (Tilley et al., 2014). Therefore, the two terms should not be used interchangeably.
[v] Deep row entrenchment may be considered as both a treatment and end-use option as it involves the burial of FS in trenches which are then covered by soil. Trees are planted in the soil and make use of the organic matter and nutrients in the buried FS (Strande et al 2014).
[vi] Although waste stabilisation ponds (WSPs) are commonly used in wastewater treatment to provide solid-liquid separation, stabilisation and pathogen reduction, the treatment of FS alone is not recommended due to its high ammonia concentration and high organic loads and solids content. WSPs are instead more useful for treating the effluent following FS dewatering in settling-thickening tanks or planted/unplanted drying beds (Strande et al 2014).
[vii] According to Tilley *et al* (2014), in wastewater treatment a free-water surface constructed wetland is most commonly used as an advanced treatment after secondary or tertiary treatment processes, as such it is considered primarily suitable for the further treatment of effluent following FS dewatering.

The primary aim of FS treatment is to ensure environmental and public health. Treatment may be divided into stages, each with its own objective(s) as mentioned previously. Influent FS is likely to contain a significant amount of garbage, particularly if it has originated from a latrine pit, and therefore requires initial screening (Strande et al., 2014). The solid waste retained on the screens requires disposal separately to the FS. Following initial screening, further treatment will normally depend on the intended end-use or disposal. For example, if sludge is intended to be used as a soil conditioner in agriculture then solid-liquid separation, stabilisation and significant pathogen reduction are vital. On the other hand, for biogas production fresh FS is required and will not require any treatment aside from screening prior to being fed into a biogas generator (or anaerobic digester) (Klingel et al., 2002). In general, the primary treatment of FS is taken to refer to further stabilisation and/or the separation of solid and liquid phases.

Solid-liquid separation, or dewatering, is important as FS can contain large volumes of water which is costly to transport and store (Strande et al., 2014). The discharge of polluted water into the environment will have significant negative impacts and dewatering is required prior to resource recovery for applications such as composting, or combustion as a fuel (Strande et al., 2014). Fresh FS is more difficult to dewater as it does not settle well and tends to foam thus it will require stabilisation prior to dewatering. This may occur using two separate technologies such as a sedimentation/thickening pond followed by an unplanted drying bed, or it may occur in one unit such an Imhoff tank. Alternatively fresh FS may be mixed with digested FS at ratios enabling quasi-discrete settling (digested FS: fresh FS 2-3:1 by volume) (Heinss et al., 1998; Koné and Peter, 2008). Dewatering is based on a number of physical mechanisms which are outlined in Table A-1. The FS treatment technologies which employ these mechanisms and their suitability for fresh and/or digested sludge are also shown. Technologies in bold are those that are relatively well-established for FS dewatering. The remaining technologies are in development and research is ongoing as to their suitability for FS dewatering. Dewatering technologies shown in Table A-1 which have not yet been trialled for FS are not shown in Table A-1.

Table A-1. Physical mechanisms for dewatering faecal sludge, the treatment technologies which employ them and their suitability for fresh or digested sludge where ABR is anaerobic baffled reactor and LaDePa is latrine dehydration and pasteurisation

Mechanism	Treatment Technology	Suitable for Fresh or Digested Sludge?
Gravity Separation	• Settling-thickening tanks	• Suitable for digested sludge or a mixture of digested and fresh sludge
	• Imhoff tank	• Suitable for fresh sludge
	• ABR[a]	• Suitable for both fresh and digested sludge
	• Sedimentation/thickening ponds	• Suitable for both fresh and digested sludge
Filtration	• Unplanted sludge drying beds with filter	• Suitable for digested sludge or a mixture of digested and fresh sludge
	• Planted sludge drying beds	• Suitable for digested sludge or a mixture of digested and fresh sludge
Evaporation	• Unplanted drying beds	• Suitable for digested sludge or a mixture of digested and fresh sludge
	• Solar drying	• Suitable for both fresh and digested sludge
Evapotranspiration	• Planted drying beds	• Suitable for digested sludge or a mixture of digested and fresh sludge
Centrifugation[b]	• Centrifuge	• Suitable for both fresh and digested sludge
Heat Drying[c]	• Thermal drying	• Suitable for both fresh and digested sludge
	• LaDePa	• Suitable for both fresh and digested sludge

[a] Gravity separation may take place in an integrated settling compartment or it may take place in a separate settler or another preceding technology, for example, existing septic tanks (Tilley et al., 2014).
[b] Centrifugation is a type of mechanical dewatering most commonly used for the solid-liquid separation of wastewater sludge, however it may also be employed for the partial removal of bound water from FS (Strande et al., 2014).
[c] Influent sludge may require preliminary dewatering prior to heat drying should the initial moisture content be high, for example, the LaDePa requires sludge to have a moisture content of between 80 – 65% (Strande et al., 2014).

Dewatering produces solid (sludge) and liquid (effluent) end products which normally require further treatment depending on the intended end-use or disposal. This is to ensure further stabilisation and/or pathogen reduction of both the effluent and sludge. Further dewatering of the sludge may also be required depending on initial treatment, for example, solid-liquid separation will occur during anaerobic digestion; however, the digestate will require further dewatering once it has been removed from the biogas reactor (Klingel et al., 2002). Other technologies from which removed sludge may require further dewatering are settling-thickening tanks, sedimentation/thickening ponds and Imhoff tanks.

According to Strande *et al* (2014) "stabilisation involves the degradation of putrifiable, readily degradable material, leaving behind more stable, less degradable organics". Stabilised sludge will generally consist of more stable, complex molecules such as cellulose and lignin as well as inorganic matter and cellular matter of microorganisms that consumed readily degradable organics (Strande et al., 2014). Stabilisation is desirable for FS treatment as it reduces the oxygen demand, produces stable and predictable characteristics, reduces odours and allows for easy storage and manipulation (Strande et al., 2014). Stabilisation can occur both as a result of aerobic treatment, for example in planted drying beds or during composting, and as a result of anaerobic treatment which occurs in anywhere there is a lack of oxygen, for example, anaerobic and facultative waste stabilisation ponds, settling-thickening tanks or biogas reactors.

Pathogen reduction is necessary to ensure safe end-use or disposal and may be achieved by a variety of mechanisms. The majority of pathogens are inactivated at temperatures above 70°C (Carrington, 2001; Feachem et al., 1983). High temperatures can be achieved during processes such as thermophilic co-composting and lime treatment (Strande et al., 2014). The lower the achievable temperature, the longer time is necessary prior to pathogen inactivation. Long sludge retention times, such as in planted drying beds, can also result in pathogen reduction as "they have limited survival time in adverse conditions" (Strande et al., 2014). The ambient temperature will impact the necessary sludge retention time as higher temperatures will result in faster inactivation. During dewatering, some pathogens may sorb or settle with the solids. It has been found that the majority of pathogens in FS deposited onto drying beds can remain in the sludge (Heinss et al., 1998). Drying sludge as in active thermal drying or passive drying beds can result in a reduction in active pathogens due to dehydration. Other mechanisms used for pathogen reduction include solar/UV radiation and pH. UV radiation inactivates a cell by damaging its nucleic acid thereby preventing the replication of microorganisms. It has been found to be very effective in against certain pathogens in wastewater (Turtoi, 2013). However, it's effectiveness can be reduced depending on the

characteristics of the wastewater (in particular its turbidity and the level of organic matter), the intensity of the radiation and the length of time pathogens are exposed to the radiation. pH control can reduce pathogens as few can survive below pH 3 or above pH 10. However, pH control may also negatively impact composting and anaerobic digestion processes. Therefore, consideration of subsequent treatment steps is required.

Effluent following sludge dewatering can be high in ammonia, salts and pathogens and may be treated in waste stabilisation ponds prior to discharge into surface waters or infiltration/reed beds (Strauss and Montangero, 2002). Treated effluent may also be used in drip irrigation or surface water irrigation provided the pathogen content has been reduced adequately (Tilley et al., 2014). Contaminant levels for effluent from unplanted drying beds has been found to be significantly lower than that originating from settling-thickening tanks for example, as the majority of pathogens tends to remain in the sludge and therefore will require less treatment (Strauss and Montangero, 2002). Effluent from planted drying beds also contains low pathogen levels and may be disposed of in a soak pit (also known as a soakaway or leach pit) if there is no end-use requirement. As the effluent percolates through the soil from the soak pit, small particles are filtered out by the soil matrix and organics may be digested by microorganisms (Tilley et al., 2014). It should be noted that soak pits are unsuitable for soils such as clays or which are rocky or hard packed due to the poor adsorptive properties. They are also unsuitable for areas where the groundwater table is high as it is recommended each pit has a depth of 1.5 m – 4.0 m with the bottom not less than 2.0 m above the water table (Tilley et al., 2014). To achieve sufficient ammonia nitrogen and total Kjeldahl nitrogen removal, Koottatep *et al* (2005) reported impounding effluent in planted drying beds for a period of 6 days. Effluent may also be stored in a receiving tank following dewatering prior to discharge into the environment. The time required for storage will depend on the characteristics of the effluent.

A.3 Recommended Faecal Sludge Treatment System for Bangladeshi Context

When considering FS treatment technologies, it is necessary to consider the local context in which the system will be deployed in terms of the state of sanitation and socio-cultural practices as well as the climatic and hydro-geologic context of the area. The site chosen for the pilot study was located 6.0 km away from the centre of Faridpur Town in the Faridpur District. There is no piped sewerage network or wastewater treatment in Faridpur, and the majority of households rely on latrine pits or septic tanks to capture their waste (Practical Action Bangladesh, 2014; Way, 2013a; Stevens et al., 2015). As a result, incoming FS is likely

to be of varying strengths and any treatment option should be robust to account for this. There are a number of factors which will influence the characteristics of FS requiring treatment including type of capture and containment technology in use, household toilet usage habits, storage duration, inflow and infiltration, collection method used and climate (Strande et al. 2014). These factors are discussed in more detail in Table A-2 with regard to the context of Faridpur

Table A-2. Factors affecting the characteristics of faecal sludge to be treated at the site of the Buro Happold Bath and WaterAid Bangladesh pilot study in Faridpur

Capture and Containment Technology
• The type of capture and containment technology used will influence the volume, strength and level of stabilisation of FS. For example, septic tanks tend to generate greater volumes of sludge than pits but the total solids content is significantly less and sludge originating from septic tanks tends to be more diluted and stabilised (Foxon, Buckley, Brouckaert, Bakare, et al., 2011; Tilley et al., 2014). • The majority of households in Faridpur have access to either a single pit latrine (42.0%) or a latrine connected to a septic tank (32.0%) (Stevens et al., 2015).
Toilet Usage
• The total solids content of FS is affected by flush water, cleansing method ('washers' versus 'wipers'), cleansing material if 'wipers', and addition of greywater to the system (Strande et al., 2014). The addition of household waste e.g. menstrual hygiene products or kitchen waste will significantly increase the total solids content of both pits and septic tanks, although less rubbish tends to be disposed of in a septic tank (Foxon, Buckley, Brouckaert, Bakare, et al., 2011). • Based on national estimates from BBS and UNICEF Bangladesh (2015;2017) it is likely that some users will have a dry toilet connected to a latrine pit whilst others will have a flush/pour flush toilet connected to either a latrine pit or septic tank. Data regarding other factors was unavailable. • It is likely that FS requiring treatment in Faridpur will vary in total solids concentration from high to low which will affect the volume of solids remaining following dewatering.
Storage Duration
• Storage duration of FS will significantly affect the characteristics. The longer FS is stored the more stabilised it will be due to the biodegradation of organic matter. • Storage duration and filling rates depend on the type of technology in use, the size of the containment system, the quality of construction, the number of users and usage habits, and inflow and infiltration (Strande et al., 2014). The filling rate of a latrine pit is also affected by the type of biological activity in it (i.e. aerobic digestion versus anaerobic digestion[a]) (Partners in Development and Polllution Research Group (PRG), 2012). • Conventional septic tanks require emptying every two to five years (Yousuf and Mahmud, 2011). However, it has been shown that desludging once per year can improve their performance by reducing the organic load (Harada et al., 2008). It has been estimated that a 2.5 m^3 pit needs emptied every 11 years (Foxon, Buckley, Brouckaert, Wood, et al., 2011). Septic tanks and pits connected to public or shared toilets will fill much more quickly and require more frequent emptying. The FS originating from these toilets tends to be less stabilised, with higher organics content and ammonium (Strande et al., 2014). • There was no available data on the ages or emptying intervals of pits and septic tanks in Faridpur. However, approximately 20% of households were reported to have new sanitation systems which did not require emptying. There were an estimated 16 public toilets alongside toilets in public institutions (i.e. schools, colleges and government buildings) which are likely to require frequent emptying (Practical Action Bangladesh, 2014). FS originating from households is likely to be more stabilised having been stored for longer as the majority rely on latrine pits or septic tanks to capture their waste.

Inflow and Infiltration
• The concentration and volume of FS is significantly influenced by the inflow and infiltration of leachate into the surrounding environment from the system and/or groundwater into the system (Strande et al., 2014). Whether or not either mechanism occurs is dependent on the design of the containment system i.e. whether they are unlined, partially lined, fully lined, connected to drainfields or soakpits, and the quality of construction (Strande et al., 2014). • Should the system be permeable to any degree then the amount and rate of inflow and infiltration is dependent on the surrounding soil properties and the level of the water table. • The water table in Faridpur District lies between 3.7 – 6.1 m below ground level (Islam et al., 2016). A pit is typically 3.0 m deep and a safe vertical distance of at least 2.0 m between the bottom of the pit and the water table is recommended for prevention of groundwater contamination (Tilley et al., 2014). Thus, there is a high likelihood of groundwater contamination due to infiltration of leachate from pits; it is also highly likely that inflow of groundwater would occur which would dilute the sludge originating from pits to some extent. • The soil in Faridpur falls under the general soil types 'calcareous dark grey floodplain soils and calcareous brown floodplain soils' (FAO, 1988; Khanam and Ahmed, 2015; Imamul Huq and Manzurul Hoque, 2012). These soils may be described as silt loams and silty clay loams which contain a moderate amount of lime (Khanam and Ahmed, 2015; Alam et al., 1993). They are generally poorly drained, have a high water holding capacity and low permeability. It is likely that, provided the pit is unlined or partially lined, inflow and infiltration of leachate from the pit and groundwater into it into the pit latrine will occur slowly. This may result in greater liquid retention within the pit which may cause a larger anaerobic digestion zone. • No estimates were found detailing the most commonly adopted design for pits or septic tanks in Faridpur. Given construction of latrines and septic tanks is often undertaken by builders from the informal sector the quality of construction is likely to vary (Strande et al., 2014). The majority of households were reported not to have a soak pit for effluent which was instead discharged directly into the environment (Practical Action Bangladesh, 2014). Approximately 38.0% of households reported the discharge of effluent into the surrounding soil. The remainder of households reported discharging effluent into nearby open land or water bodies, drains and other places (Practical Action Bangladesh, 2014).
Collection Method
• It is often the case that FS removed manually is more concentrated with high viscosity than FS that has been removed by pumping. Sludge at the bottom of a containment system tends to be very think and will only be removed if it is done manually with shovels, or with the addition of water to decrease the viscosity and enable pumping (Strande et al., 2014). Septic tank sludge may also be more dilute if the majority of what is collected is supernatant instead of sludge. This may occur because the pump is not strong enough to remove all of the accumulated sludge or there are accessibility issues (Practical Action Bangladesh, 2014). • The majority of FS collection in Faridpur is done manually by sweepers with small pumps, shovels and containers (Practical Action Bangladesh, 2014). Often only the liquid or semi-solid contents of the containment system are removed so it is likely that an FS requiring treatment will be relatively dilute and viscous.

Climate
• Warmer temperatures increase the rates of biological degradation within a containment system and wet climates are likely to experience the overflowing and flooding of OSS. • Bangladesh is one of the wettest countries in the word and has a warm, humid climate influenced primarily by monsoon circulations (World Bank, 2020a). The climate in Faridpur in keeping with the rest of the country. • Although it is located on the floodplain of the River Padma, Faridpur is not considered at risk of annual flooding; however, flood events have previously been recorded in 1988, 1998 and 2004 (Practical Action Bangladesh, 2014). However, it does experience significant annual rainfall, as such OSS may be at risk of flooding and overflowing.

[a] During aerobic digestion, 50-70% of the organic matter in FS becomes biomass, while during anaerobic digestion only 5-10% does resulting a slower rate of sludge accumulation as a greater proportion of the organic matter is transformed (Partners in Development and Polllution Research Group (PRG), 2012).

When designing a FS treatment technology, it is normally desirable to produce a useful end product to 'close the nutrient loop'. Agriculture is significant in Bangladesh and "plays a dominant role in the growth and stability of the economy" (FAO, 2014a). It is the largest employment sector in the country and contributes approximately 16.5% of the country's Gross Domestic Product (GDP) (CIAT and World Bank, 2017). The agricultural productivity of soils in Faridpur has been found to range between moderate and poor (Imamul Huq and Shoaib, 2013). Treated FS may be used as a soil conditioner and has been found to have many benefits over the use of chemical fertilisers alone. As noted in Strande *et al* (2014) the "organic matter in FS can increase soil water holding capacity, build structure, reduce erosion and provide a source of slowly released nutrients." Therefore, all of the treatment technologies considered allowed for the production of compost which would be safe for use in agriculture. Based on the desired end-use, site visits in Bangladesh to establish the context of FSM in the country and the brief provided by WaterAid Bangladesh, the following treatment options were considered for the pilot project: co-composting with organic solid waste; planted drying beds; small-scale biogas generation (i.e. anaerobic digestion); and unplanted drying beds (Way, 2013a)[5]. A comparison of the four treatment technologies is shown in Table A-3.

[5] Pit additives were also considered; however, they have previously been proven to be largely ineffective (Buckley et al., 2008). As such they are not discussed in this thesis, but discussion of the advantages/disadvantages may be found in Way (2013a).

Table A-3. Comparison of faecal sludge treatment options for pilot study in Bangladesh (adapted from (Way, 2012)*)*

Criteria	Co-Composting with Organic Waste	Small-scale Biogas Generation	Planted Drying Bed	Unplanted Drying Bed
Treatment Process	Secondary	Primary	Combined	Primary
Adapted for wet climate?	Yes, if covered	Yes	No	No
Land area requirement	Large, especially if open system is used	Small, majority of structure can be built underground	Large	Large
Can use local materials?	Yes	Yes – if constructed from bricks rather than a refabricated tank	Yes	Yes
Investment Cost	Low	Moderate	Moderate	Low
Operation Cost	Low	Low	Low	Low
Construction, Operation and Maintenance	Requires expert design and operation by skilled personnel	Requires expert design and skilled construction and operation	Requires expert design and construction and trained operators; sludge removal is labour intensive	Simple operation but requires trained operators and expert design and construction; sludge removal is labour intensive
Odour	Some	No	Can be significant	Can be significant
Sludge storage time	6 – 8 weeks (at least)	Continuous removal	3 – 5 years	10 – 15 days (depending on climatic conditions)
Pathogen removal	High	Limited depending on hydraulic retention time, digestate may require further treatment to be safe for end-use	High in dewatered sludge but leachate requires further treatment to be safe for end-use	Limited, dewatered sludge and leachate require further treatment to be safe for end-use
Useful end product?	Yes - Compost	Yes – biogas and compost (following further treatment of digestate)	Yes – forage, compost and water for agriculture or aquaculture (following treatment of leachate)	Yes – compost and water for agriculture or aquaculture following further treatment

With low investment and operation costs, co-composting may achieve high pathogen reduction and "provides a valuable resource that can improve local agriculture and food production" (Tilley et al., 2014). However, it requires a large land area and long storage times as well as being labour intensive. Additionally, co-composting is only suitable for drier sludge as the optimum moisture content for the process sits between 40-60% by weight. If the moisture content is any higher than 60% it can impede microbial growth by creating anaerobic conditions (Strande et al., 2014). Wetter sludge will require dewatering prior to undergoing co-composting.

In contrast to co-composting, the consistency and moisture content of FS has little bearing on the function of planted or unplanted drying beds. That said, fresh FS requires some stabilisation prior to treatment on drying beds, or it should be mixed with stabilised sludge. Drying beds can be locally constructed and require no electrical energy. Investment costs are moderate for planted drying beds and low for unplanted drying beds while operation costs are low for both technologies. The key advantage of planted drying beds over unplanted drying beds it that "the root and rhizome system of the plants used… creates a porous structure in the layer of accumulated solids, thus enabling [the maintenance of] the dewatering capacity of the filter for several years" (Koné and Peter, 2008). Additionally, the prolonged sludge storage time on planted drying beds will result in greater stabilisation while sludge treated on unplanted drying beds will require further treatment following removal. Predation, dehydration and retention time all contribute to significant pathogen reduction on planted drying beds. In contrast the main purpose of an unplanted drying bed is dewatering as such, sludge storage time tends to be much shorter and there is limited pathogen reduction and stabilisation (although both processes may occur). Therefore, the sludge will require further treatment upon removal from the drying bed. Disadvantages to both planted and unplanted drying beds include the large land area requirement, odours and flies may be problematic and they require expert design and construction (Tilley et al., 2014). Additionally, drying beds are not adapted to a rainy climate or areas prone to flooding as are common in Bangladesh.

Although requiring less land and generating fuel as well as being less susceptible to flooding, small-scale biogas generation was also discounted as a potential FS treatment to fill the brief given by WaterAid Bangladesh. For successful operation a biogas reactor requires expert design and skilled construction and initial costs are relatively high. Reactors are only suitable for the treatment of fresh FS and require a regular flow of sludge. Thus, they are most suited for sludge from public toilets as pits and septic tanks are emptied as a matter of routine or they are directly connected to private or public toilets. The digestate is likely to require

further treatment due to incomplete pathogen removal and its removal can cause difficulties (Tilley et al., 2014; Way, 2013a). Furthermore, a report by Practical Action Bangladesh (2014) noted that an anaerobic digester had previously been constructed in 2009 at Baridpur, Faridpur. However, the project did not progress due to objections from local households regarding the treatment of FS at the site and technical difficulties with depositing sludge from Vacutugs into the digester.

Based on the comparison of the different potential treatment technologies, the brief provided by WaterAid Bangladesh and the large solar resource in Bangladesh, the team at BH elected to adapt the design of unplanted drying beds to the Bangladeshi context. This involved building the drying beds above ground and providing a roof both to prevent rain entering the drying beds and in an attempt to provide a 'greenhouse effect' as used for solar drying of sludge. To address the significant land area requirement, the drying beds were reduced in size (1.5 m x 3.0 m) to "suit batch delivery of (comparatively) small volumes of sludge, reduce construction costs and enable modular construction to meet demand of the particular location" (Way, 2013b). Additionally, the smaller size of the drying beds would allow for easier manual removal of dried sludge. The final design for the covered, unplanted sludge drying beds is shown in Figure A-2. The drying beds were low cost[6] with low energy requirements and could be constructed from locally available materials.

[6] Although it is difficult to provide a precise cost analysis as it can vary dependant on the cost of local materials, it was estimated that the cost of construction for one drying bed to treat 900 L of FS in 5-10 days would be between 55000.00 – 60000.00 BDT (700.00 – 770.00 USD) and an associated reedbed for effluent treatment would cost between 100000.00 – 110000.00 BDT (1284.00 – 1412.00 USD) (Way, 2013b).

Figure A-2. Final design of covered, unplanted sludge drying bed for Buro Happold Engineering Bath and WaterAid Bangladesh pilot project

The BH Bath pilot study was conducted between November 2012 and September 2013, encompassing monsoon season. Upon completion of the pilot study it was concluded that liquid waste could be dried to a consistency suitable for co-composting in a short period of time (ten days) on a covered, unplanted drying bed with a filter bed, even during monsoon season. It was found that a polythene tunnel, which included zippered end closures to alter ventilation, was the best performing roof material. The transparent material "increased the heat gain of the sludge, and allowed greater UV penetration, which is linked to improved pathogen reduction" (Way, 2013a). Drying sludge in a fully enclosed bed led to higher heat retention within the bed at night but little difference during the day. The roof itself appeared to have been of most use to keep rainfall off the sludge rather than in increasing temperatures within the bed or increasing drying rates. Seck *et al* (2015) drew a similar conclusion when assessing the drying rates of FS in greenhouses with sand and gravel filters. Raking the sludge on the beds was found to minimally improve drying and therefore was recommended but was not considered necessary to ensure adequate performance of the drying bed. Prior to being deposited onto the drying beds, initial screening was recommended as it was found that influent sludge contained significant solid waste. No initial solid-liquid separation treatment was recommended as it was found that the influent sludge was largely stabilised and therefore dewatered well. Should influent sludge be a mixture of fresh and stabilised sludge an initial holding tank may be recommended for mixing prior to the FS being deposited onto the covered, unplanted drying beds. Following dewatering on the drying beds, sludge was

removed and co-composted with woody organics at a ratio of 1:2, while the liquid effluent was treated in a reedbed (a simple constructed wetland) adjacent to the drying beds prior to discharge into a local watercourse or storage in a pond for the maintenance of the compost moisture content. The recommended FS treatment process is shown in Figure A-3.

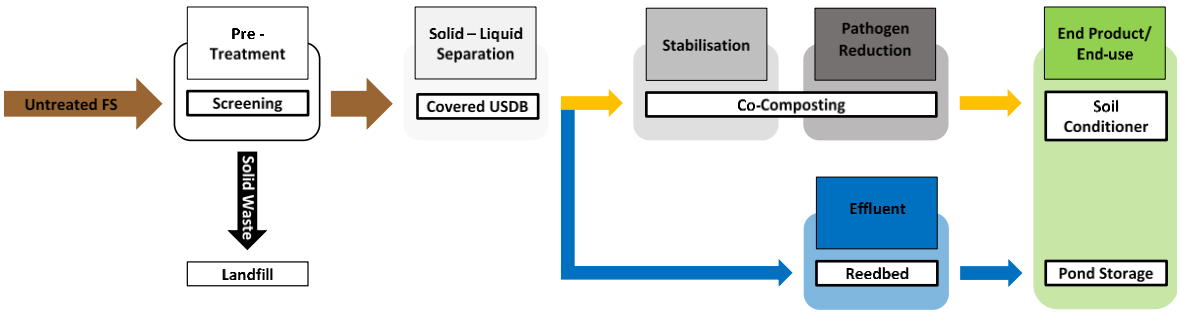


Figure A-3. Recommended process for faecal sludge treatment for Buro Happold Bath and WaterAid Bangladesh pilot project where FS is faecal sludge, the black arrow is indicative of solid waste, the yellow arrow is indicative of sludge and the blue arrow is indicative of effluent following dewatering

Thus, if properly designed and operated, a covered USDB with a filter bed can produce a solid product suitable for use as a soil conditioner or which can be disposed of into the environment in designated areas without adverse impacts. This is provided dried sludge is subject to a period of further storage and composting to stabilise and reduce pathogen content to an acceptable level.

B Measures of Wastewater Sludge Dewaterability

There are two main tests used to measure the dewaterability of sludge: specific resistance to filtration (SRF) and capillary suction time (CST). SRF is "a measure of the resistance offered by a sludge to the withdrawal of water" (Karr and Keinath, 1978). The higher the SRF the more difficult a sludge is to dewater. It is based on the analysis of pressure drop for flow through a porous medium using Darcy's Law resulting in a sludge characterisation parameter related to permeability (Vesilind, 1988). As outlined in Smollen (1986) SRF is determined by dewatering a sludge sample by filtration through a filter paper (usually Whatman No 1 and 541) in a Buchner funnel under a vacuum differential or pressure differential. Volume of filtrate collected, V (m^3), and time, t (s), are recorded and a graph of t/V versus V is plotted. The slope of the straight line, b, is substituted into the specific resistance equation:

$$r = \frac{2bPA^2}{\mu C}, \qquad (B1)$$

where r (cm g^{-1}) is the specific resistance to filtration, P (Pa) is the applied pressure differential, A (cm^2) is the filter area, μ (Pa s) is the filtrate viscosity and C (g cm^{-3}) is the solids concentration of the sludge (Smollen, 1986; Karr and Keinath, 1978).

There are significant limitations the usefulness of SRF in describing sludge dewaterability. SRF theory, as described by Equation B1, assumes that a thicker sludge cake (i.e. the dewatered sludge) will be deposited as the sludge solids concentration increases (Karr and Keinath, 1978). This leads to increased resistance to filtration which is accounted for by an increase in b of Equation B1 so that the increase in solids concentration is offset. This leads to a value of specific resistance to filtration that is independent of solids concentration (Karr and Keinath, 1978). However, Karr and Keinath (1978) found that SRF is dependent on the sludge solids concentration "when the concentration of supra-colloidal solids is sufficient to cause blinding[7]...[which] reduces the sensitivity of b to changes in cake thickness because filtration resistance caused by blinding is of a greater magnitude than that caused by changes in cake thickness." Increase in SRF as a result of blinding has been commonly observed in sludges with a high degree of dispersion and low ESP content (Mikkelsen and Keiding, 2002).

Another limitation to SRF theory is the assumption of constant mean porosity, constant mean cake size, and constant pore water pressure when this is not the case in reality (Sawalha and

[7] Cake blinding is used to describe the entrapment of small particles within sludge cake pores in suspensions with a large particle size distribution (Christensen et al., 2015). It may also occur because of floc degradation.

Scholz, 2010). Porosity and pore-water pressure vary across the sludge cake and the size of the cake changes during testing with a Buchner funnel, especially at the beginning and end, when the suspended solids concentrations change rapidly (Sawalha and Scholz, 2010). The SRF test also doesn't account for all of the properties of a sludge known to affect dewatering, for example, "pick up and release properties, scrolling properties, and particle resistance to shear," (Karr and Keinath, 1978). Although it seems unlikely that any test would account for all properties. Results are also sensitive to the specifics of the test procedure and therefore are difficult to reproduce. Additionally, performing the SRF test is "time consuming and cumbersome" (Vesilind, 1988) as well as expensive (Gray, 2014).

The CST was developed by Gale and Baskerville (1967) as a simpler method for determining sludge dewaterability and evaluating the effects of pre-treatment chemicals on sludge filterability (Gray, 2014). CST apparatus is set up as shown in Figure B-1. Sludge is placed in the reservoir which sits on absorbent filter paper. Water is drawn from the sludge by the capillary suction of the filter paper and CST is measured by noting the time taken for the water to travel a set distance across the paper using two electrodes placed at specific intervals. Sludges with a low CST will dewater more easily than those with a high CST. The rate at which the water moves is determined by "the properties of the paper, the viscosity of the filtrate and the filterability of the sludge" (Smollen, 1990).

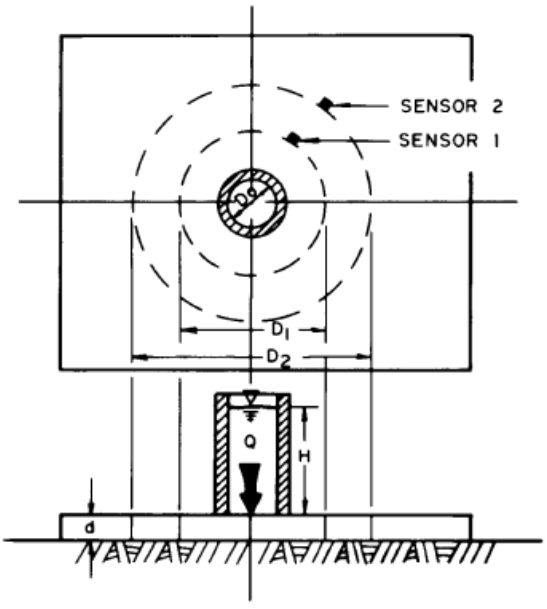


Figure B-1. Schematic diagram of capillary suction time apparatus (Vesilind, 1988)

Vesilind (1988) developed a mathematical model for CST based on Darcy's Law and a filterability constant. For clean filter paper and pure water, CST can be found from:

$$t = (D_2^2 - D_1^2)\left(\frac{\pi d}{Q}\right), \qquad (B2)$$

where t (s) is the capillary suction time, D_1 and D_2 (cm) are the diameters at which the two sensors in the CST apparatus are located as shown in Figure 3-13, d (cm) is the thickness of the filter paper, and Q ($cm^3\ s^{-1}$) is the flow rate from the collar into the filter paper. For clean paper and pure water Q is constant; however, over the course of testing a sludge sample, solids accumulate on the surface of the filter paper resulting in a reduction in Q. Assuming the effect of thickness of the sludge sample in the collar is negligible, then for flow through the sludge deposited on the surface of the filter paper Darcy's Law may be written as follows:

$$Q = \frac{AkP}{\mu S}, \qquad (B3)$$

where A (cm^2) is the area of the bottom of the collar, k (cm^2) is the permeability, P (kPa) is the capillary suction, μ (Pa s) is the viscosity and S (cm) is the thickness of solids deposited on the surface of the filter paper. Assuming that the deposited cake thickness is directly proportional to the sludge solids concentration, C, and replacing k with the filterability constant and combining Equation B2 and B3 then:

$$t = \left[(D_2^2 - D_1^2)\left(\frac{\pi d}{AP}\right)\right] \cdot \left[\frac{\mu C}{\chi}\right], \qquad (B4)$$

where C solids concentration and χ is the filterability constant. The first term of Equation 3.38 relates to the test apparatus and the last term describes the sludge sample: viscosity, solids concentration and filterability (Vesilind, 1988). The filterability constant incorporates permeability and the conversion factor from thickness, S, to solids concentration, C, for a given surface area, as well as "unmeasurable sludge properties such as particle size distribution, surface chemistry, and so on" (Vesilind, 1988).

In a review of literature on CST, Scholz (2005) found that there was a significant correlation between filtrate viscosity and CST. According to the Vesilind (1988) model, CST should vary linearly with viscosity; by varying the temperature of the sludge tested, Vesilind (1988) found this held true up to approximately 65°C at which point CST levels off and then increases, as shown in Figure B-2. This is because above this temperature cells in the sludge are being lysed – the cell membranes are broken down – which significantly affects particle size distribution in the sludge and increases CST. That said, Sawalha and Scholz (2012) observed a non-linear relationship between all temperature values and CST values, likely due to the "complex effects temperature has on sludge viscosity, filterability, settleability, desorptivity and flocculation behaviour" (Gray, 2014).

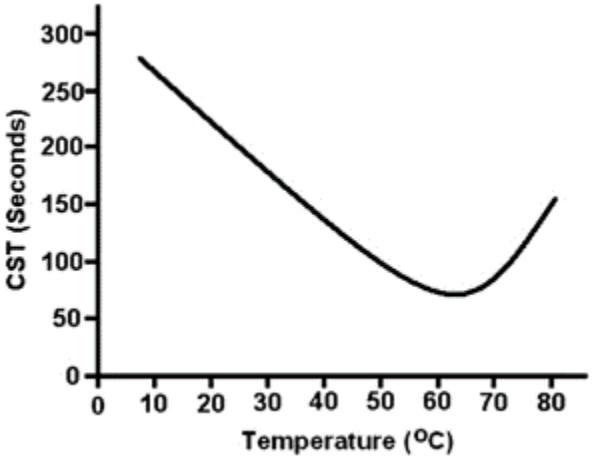


Figure B-2. Variation of capillary suction time with temperature (Vesilind, 1988)

As with SRF there are a number of limitations to the usage of CST. It is not a fundamental measure of dewaterability and is a measurement unique to a specific sludge solids concentration, the ambient temperature of the test and the apparatus – such as type of filter paper – used (Vesilind, 1988). Therefore, results cannot, or should not, be compared between different wastewater treatment plants (Gray, 2014). Sawalha and Scholz (2007)[8] found that CST values were significantly influenced by the type of filter paper used, sludge reservoir (collar) geometry and type of sludge. Another drawback of CST theory is that it doesn't take into account the effect of sedimentation which may then lead to an overestimation of sludge resistance to water removal (Scholz, 2005). Scholz (2005) suggested the application of a rectangular sludge reservoir to solve the problem. According to Jin *et al* (2004) the issue with CST is that it "does not quantify a particular, fundamentally based physical parameter of the sludge." Raynaud *et al* (2012) noted that CST should be considered more as an "empirical tool for engineering practices to optimise flocculation than as a test allowing to a careful analysis of dewatering devices." Normalised CST, found by dividing CST values by the initial total suspended solids content or total solids content, as recommended by Pen *et al* (2011) may be more useful for comparisons but is still susceptible to the limitations discussed previously. In terms of water classification, Chen *et al* (1996) concluded that CST is "a good index for the product of solid concentration and average specific resistance, however, it cannot be directly used to evaluate bound water in the sludge" (Chen et al., 1996). Similarly, Smollen (1990) found that there was no apparent correlation between either SRF or CST and intracellular and interstitial water but CST may be related to volume of free, unbound water (Jin et al., 2004).

[8] Sawalha and Scholz (2007) ran tests on primary sludge, activated sludge, secondary sludge, gully pot sludge and a synthetic sludge developed for benchmarking purposes.

Further limitations to both SRF and CST relate to the impact of solids content: minor changes in initial solids concentration has been observed to be critical to reproducibility (Skinner et al., 2015). This is particularly true for sludges with a low initial solids content, in the region of 2.0% and 10.0% total volume. It is also worth noting that CST and SRF focus on the initial stage of wastewater sludge dewatering i.e. filtration (Skinner et al., 2015; Novak, 2007; Raynaud et al., 2012). Where a description of the dewatering of sludge over the full dewatering process is required it is best to use other parameters, such as those outlined by Skinner *et al* (2015).

According to Skinner *et al* (2015) quantification and comparison of the dewaterability of different sludges using SRF and CST is difficult given the widely varying compressibility and large range in permeability. For the whole of the dewatering process, the dewatering behaviour of a sludge may be related to the solids diffusivity, $D(\varphi)$, defined as:

$$D(\varphi) = \frac{dP_y(\varphi)}{d\varphi} \cdot \frac{(1-\varphi)^2}{R(\varphi)}, \qquad (B5)$$

where φ is the volume fraction of solids, $P_y(\varphi)$ is the compressive yield strength of the suspension and $R(\varphi)$ is the hindered settling function which is inversely proportional to the permeability (Skinner et al., 2015). Following the input of parameters into a filtration model, quantitative comparison was possible between sludges at set values of final average solids concentration or specific solids throughput. However, it was recommended that "a statement that one sludge dewaters better than another needs to be made with care and the solids concentration of comparison specified" (Skinner et al., 2015). Christensen *et al* (2015) recommended the use of specific filtration flow (SFF) rate to "characterise the liquid flow through a cake with high compressibility[9]" and encompass both the filtration and consolidation stages of dewatering. SFF may be found from:

$$SFF = q\omega_c = \frac{p}{\mu\alpha_{av}}, \qquad (B6)$$

where q ($m^3\ m^{-2}s^{-1}$) is the filtrate flux, ω_c ($kg\ m^{-2}$) is the mass of solid particles per unit filter media area, p (Pa) is the filtration pressure, μ ($N\ s\ m^{-2}$) is the filtrate viscosity and α_{av} ($m\ kg^{-1}$) is the average specific resistance of the cake (Christensen et al., 2015). For highly compressible cakes, such as those from activated sludge, the SFF value is constant and independent of pressure assuming that an increase in pressure is accounted for by the subsequent increase in specific resistance to filtration (Christensen et al., 2015). Therefore,

[9] A cake is deemed compressible if porosity decreases with increasing pressure which results in higher resistance to filtration (Christensen et al., 2015).

SFF may be a better measure of dewaterability compared to CST or SRF. It can be used to compare the dewaterability of different wastewater sludges except at low pressure (Christensen et al., 2015). Raynaud *et al* (2012) recommend the use of a filtration-compression scale to determine sludge dewaterability over the full dewatering phase.

Another alternative measurement is the filterability constant, χ. Like SRF, χ is a fundamental measure of dewaterability and is relatively simple to determine. Rearranging Equation 3.38 for χ and for a given set of tests, using common apparatus and filter paper then:

$$\chi = \Phi\left[\frac{\mu C}{t}\right], \qquad (B7)$$

where Φ is a dimensionless instrument constant that shouldn't vary with the sludge sample used and all other terms are as defined previously. Veslind (1988) found that χ remained relatively constant even when there were large variations in viscosity or sludge solids concentration. Filterability is inversely proportional to SRF, therefore a sludge which has a low χ value is difficult to dewater whilst a sludge which has a high χ value is easy to dewater. According to Vesilind (1988) χ may be used as a general descriptor of the rate of dewatering of a sludge; however, it has not been found to be reported often in the literature.

Other measures of dewaterability include: the bound water content; sludge volume index (SVI); cake dry matter content; and turbidity of filtrate, as described in Raynaud *et al* (2012). Turbidity is an indication of the proportion of suspended solids in a sample which are known to impact dewaterability. Cake dry matter content is a measure of the extent of dewaterability. SVI is often reported in studies of sludge dewaterability (Sobeck and Higgins, 2002). It is commonly used as a measure of sludge settleability but Dick and Vesilind (1969) note that it "defines only one point on the settling curve, it is not a precise measure of settling characteristics." It has also been used to infer bound water content, assuming that water trapped in the settled sludge equates to the bound water (Vesilind and Hsu, 1997). Dick and Vesilind (1969) found that the SVI for activated sludge was highly dependent on suspended solids concentration but practitioners commonly compared SVI values ignoring solids concentration. This is similar to the limitation Skinner *et al* (2015) noted with regards to the comparison of CST values and SRF values for sludge. Dick and Vesilind (1969) also found that SVI was considerably influenced by temperature and cylinder diameter as well as by stirring to some extent. They concluded that SVI is "a very nonspecific, arbitrary measure of the physical characteristics of activated sludge...[that] is not related consistently to any basic physical property" (Dick and Vesilind, 1969). The relative influence of different parameters varies between different sludges as well as between different concentrations of the same sludge. Therefore, SVI comparisons of different sludges, or different concentrations of the

same sludge, may be relatively meaningless even when conducted according to a standard method.

A number of measures of dewaterability are available, the most commonly used are SRF and CST. It is important to recall the limitations when comparing reported data based on the measures of dewaterability and extent of dewaterability for different sludges. Comparisons of the dewaterability of sludges should be made with caution.

C Simulants Developed for Fresh Faeces

The key work on the development of faecal simulants has been conducted by NASA for the specific purpose of treatment of waste in space systems. According to Kaba *et al* (1990) the electrochemical incineration of human waste is of particular relevance in space vehicles where treatments which convert waste into recyclable chemicals is of particular importance during long missions. To investigate the proposed treatment method, Kaba *et al* (1990) developed a simulant based on the assumption that human faeces consisted of one third microorganisms and intestinal flora, one third undigested fibre and the final third being composed of lipids and inorganic materials. The simulant is shown in Table C-1, it was designed to represent many of the constituents of faeces (Kaba et al., 1990). Cellulose was used to represent the indigestible component of faecal waste. This may have been based on the fact that the carbohydrate fraction of faecal matter consists primarily of undigested cellulose, vegetable fibres and pentosan (Rose et al., 2015). Dried Torpulina and Escherichia coli (E.coli) were used to represent the microbial content and oleic acid was used to represent the fats (lipids) content (Bockris et al., 1988; Kaba et al., 1990). Casein[10] provided the protein content and the rest of the components represented inorganic material (Kaba et al., 1990; Bockris et al., 1988). Kaba *et al* (1990) noted that the simulant had a paste-like consistency and gave off a strong odour. No other comparison was made between the simulant and real faeces. As the simulant was representative of the chemical composition of real human faeces it was assumed to behave in a similar manner to real faeces.

Table C-1. Composition of synthetic faeces were M (kg) is mass, KCl is potassium chloride, NaCl is sodium chloride and $CaCl_2$ is calcium chloride (Kaba et al., 1990)

Component	M (kg)	% total dry M
Cellulose	0.60	33.0
Torpulina	0.43	25.0
E.Coli	0.12	7.0
Casein	0.17	0.10
Oleic acid	0.37	20.0
KCl	0.04	2.0
NaCl	0.04	2.0
$CaCl_2$	0.03	1.0
TOTAL	**1.80**	**100**

[10] Casein is a phosphoprotein from bovine milk (Sigma-Adrich, n.d.).

To investigate resource recovery following treatment of waste by incineration, Fisher *et al* (1998) simplified the recipe developed by Kaba *et al* (1990) as shown in Table C-2. Yeast was used in the place of E.Coli and Torpulina to represent the microbial content and oleic acid was replaced with peanut oil on the basis that the main fatty acid in peanut oil is oleic acid. No rationale was given for the elimination of casein or sodium chloride and addition of calcium phosphate in the recipe.

Table C-2. Composition of modified synthetic faeces were M (kg) is mass, KCl is potassium chloride and $Ca(H_2PO_4)_2$ is calcium dihydrogen phosphate (Fisher et al., 1998)

Component	M (kg)	% total dry M
Cellulose	0.38	37.5
Yeast	0.38	37.5
Peanut oil	0.20	20.0
KCl	0.04	4.0
$Ca(H_2PO_4)_2$	0.01	1.0
Water	1.5	
TOTAL	**2.51**	**100.00**

The amount of water in the simulant developed by Fisher *et al* (1998) was found to have a moisture content of 61% which is relatively low as faeces has been found to have a median value of 75% water with a range of between 63% and 86% (Wignarajah et al., 2006; Rose et al., 2015). Based on the median moisture content of 75%, then remaining 25% of faeces is composed of solids. According to Rose *et al* (2015) organic matter makes up between 84% and 93% of this solid fraction; the remainder being composed on non-organic constituents. Table C-3 shows the major components of the solid material of human faeces based on chemical composition[11].

[11] Fractions of components are highly dependent on dietary intake and biological availability (Rose et al. 2015).

Table C-3. Major components of the solid material of human faeces where DS (%) is dry solids found in the literature

Component	Fraction DS (%)
Bacterial biomass[12]	25.0 – 54.0[a]
Nitrogen[13]	5.0 – 7.0[a]
Protein	2.0 – 3.0[b]
Carbohydrate or other non-nitrogenous undigested plant matter (fibre)	10.0 – 30.0[c]
Lipids	10.0 – 20.0[b]
Minerals: Ca	4.5[d]
Minerals: PO_4	3.0 – 5.4[d]
Minerals: K	1.0 – 2.5[d]

[a] Rose *et al* (2015)
[b] Guyton and Hall (2011)
[c] Wignarajah *et al* (2006)
[d] Feachem *et al* (1983)

According to Wignarajah *et al* (2003) soluble elements such as sodium are almost completely absorbed in the gastrointestinal tract whilst other elements such as calcium and phosphate are less readily absorbed, so a greater fraction of that ingested is released in faeces. This would give a likely reason for why Fisher *et al* (1998) eliminated sodium chloride from the recipe developed by Kaba *et al* (1998). Table C-3 shows that the main minerals excreted in faeces are calcium, phosphate and potassium, this is the likely reason that calcium phosphate was included in the recipe developed by Fisher *et al* (1998). From Table C-3 it is clear that the protein content of faeces is low, and this may be why Fisher *et al* (1998) eliminated casein from the recipe developed by Kaba *et al* (1998). Fisher *et al* (1998) found that the synthetic faeces ash after burning compared well to data for real faeces reported in the literature.

Wignarajah *et al* (2006) developed simulants to reproduce the water retention and rheological characteristics of faeces, building on the work done by Kaba *et al* (1990) and Fisher *et al* (1998). It is known that volume, composition and consistency of faeces depends on factors such as diet, climate and health of the individual (Feachem et al., 1983). Therefore, the recommended simulants developed by Wignarajah *et al* (2006) were based on the key assumption that, under controlled conditions with healthy adults (e.g. astronauts) fed a standard diet, normal variation in physical and chemical composition of faeces and urine

[12] Wide variation has been found due to differing methodologies used to calculate bacterial biomass (Rose et al., 2015; Wignarajah et al., 2006).
[13] According to Rose *et al* (2015) nitrogen in faeces is also recorded as protein; fraction of protein can be estimated from nitrogen using the nitrogen-to-protein conversion factor of 6.25.

would be limited (Wignarajah et al., 2006). Wignarajah *et al* (2006) considered the elemental composition of human faeces and established the following empirical formula: $C_1H_{1.87}O_{1.11}N_{0.2}$. Five combinations of simulant were developed based on this formula as shown in Table C-4

Table C-4. Different combinations of chemicals to represent human faeces simulants (Wignarajah *et al*, 2006)

Component	%Wt Comb 1	%Wt Comb 2	%Wt Comb 3	%Wt Comb 4	%Wt Comb 5
E.Coli	30.0	30.0	30.0	30.0	30.0
Cellulose	0.0	15.0	15.0	0.0	10.0
Polyethylene glycol	20.0	20.0	20.0	10.0	5.0
Psyllium	20.0	5.0	0.0	5.0	0.0
Peanut oil	20.0	20.0	20.0	20.0	20.0
Miso	5.0	5.0	10.0	30.0	30.0
Inorganics	5.0	5.0	5.0	5.0	5.0
Dried coarsely ground vegetable matter	50.0 mg	50.0 mg	50.0 mg	50.0 mg	50.0 mg

Yeo and Whelchel (1994) developed a faecal simulant to be used in the development of "personal care devices such as diapers, training pants and incontinence garments." The primary aim was for the simulant to replicate the dewatering behaviour of real faeces, specifically its water binding ability. Yeo and Whelchel (1994) recommended a simulant consisting of 80% by total weight water, 15% by total weight polyvinyl-pyrrolidone and approximately 5% by total weight psyllium hydrophilic muccilloid. Although the simulant developed by Yeo and Whelchel (1994) represented real faeces in terms of dewatering rate and consistency, Wignarajah *et al* (2006) noted that there was no way to monitor microbial activity after processing. Furthermore, the use of polyvinyl-pyrrolidone resulted in a much higher level of nitrogen than that typically found in human faeces. Wignarajah *et al* (2006) opted to use non-nitrogen containing polyethylene glycol (PEG) to represent the water-holding capacity of faeces and included psyllium in their recipe. Miso paste was included likely due to its composition – 38% protein, 21% fats, 20% fibre and 4% minerals – which meant it could represent a number of components of real faeces as well as contributing to the colour of the simulant. Simulants two and three were found to be lighter in colour given the reduced psyllium content whilst combinations four and five were more similar in colour to real faeces due to the higher miso content. Wignarajah *et al* (2006) concluded that the simulants were chemically and physically analogous to real human faeces based on the elemental composition of the simulants compared to real faeces but recommended further tests for rheological characteristics and dewatering behaviour.

To test the performance of a microwave enhanced freeze drying solid waste processor for water recovery from waste, Wignarajah *et al* (2008) used recipe five from Table C-4 The recipe was adapted slightly by removing the bacterial component for simplicity and safety reasons (Wignarajah et al., 2008). The inorganics used were potassium chloride (4.0% of total mass) and calcium chloride (1.0% of total mass). The primary interest of the testing was the total percentage water removal achieved in simulants of moisture contents varying between 40% and 90%. It was found that up to 95% of water could be removed within 45 minutes and little difference was observed between high and low water contents. There was no comparison made between water removal of synthetic faeces and real faeces.

Thermochemical carbonisation has become an area of increasing interest in the treatment of human waste. The process is distinct from incineration as it occurs in the absence of oxygen. Where treatment occurs in the presence of water the process is termed hydrothermal carbonisation whilst in the absence of water, it is known as pyrolysis (Paneque et al., 2017). Danso-Boateng *et al* (2013) used the simulant proposed by Fisher *et al* (1998) in decomposition kinetics in a hydrothermal batch reactor as part of the Bill and Melinda Gates Foundation 'Reinvent the Toilet Challenge[14].' Danso-Boateng *et al* (2015) established kinetics of hydrochar production and decomposition of primary sewage sludge and synthetic faeces. Table C-5 shows the characteristics of the influent primary sewage sludge; the same data were not reported for the simulant used. However, assuming that the simulant replicated the elemental composition of real faeces as given by Fisher et al (1998) in Table C-6 then it appears to compare well with primary sewage sludge and may be expected to behave in a similar manner.

[14] In 2011, the Water, Sanitation and Hygiene program of the Bill and Melinda Gates Foundation initiated the 'Reinvent the Toilet Challenge.' The aim was to "generate and support new approaches for toilet technologies that safely and effectively manage human waste" (Bill and Melinda Gates Foundation, 2019).

Table C-5. Characteristics of influent primary wastewater sludge where C is carbon, HHV is higher heat value, H is hydrogen, O is oxygen, N is nitrogen, TOC (g^{-1}) is total organic carbon, BOD (g^{-1}) is biochemical oxygen demand and COD (g^{-1}) is chemical oxygen demand (Danso-Boateng et al., 2013)

Property	**Feedstock Dried**
Proximate analyses	
Moisture (%)	8.73
Ash (% db)	15.97
Volatile matter (% db)	70.34
Fixed C[a] (% db)	5.33
Calorific value (HHV)	18.01
Ultimate analyses	
C (% db)	46.93
H (% db)	6.11
O[b] (% db)	50.90
N (% db)	4.14
Liquid filtrate	
TOC ($g\ L^{-1}$)	2.17
COD ($g\ L^{-1}$)	36.60
BOD ($g\ L^{-1}$)	7.01

[a] 100 – (moisture + ash + volatile matter)
[b] 100 – total C/H/N

Table C-6. Elemental composition of faeces where C is carbon, H is oxygen, N is nitrogen, O is oxygen, S is sulphur, M_{db} (%) is mass dry basis and NR is not reported (Fisher et al., 1998)

Component	**M_{db} (%)**
C	43.0
H	6.3
N	2.4
O	NR
S	NR
Elemental ash	10.0 – 15.0

It was found that a higher temperature was necessary to initiate solids decomposition in the simulant sludge to produce hydrochar and the activation energy for primary sludge was lower than that for the simulant. The decomposition of both the simulant and primary sludge follow first order reaction kinetics. No comparison was drawn between the composition of the simulant and primary sludge hydrochar following treatment and no conclusion was drawn as to whether the simulant was representative of the real sludge. Afolabi *et al* (2014) also used a synthetic simulant to test hydrothermal treatment. The authors do not provide the specifics of the recipe but reference Wignarajah *et al* (2006) and therefore it is likely to be one of the

five simulant recipes shown in Table C-4 Sewage sludge, excreta (mixture of urine and faeces) and faecal sludge were all also tested. Physical properties (smell, colour and texture), dewaterability – measured by capillary suction time (CST) – and calorific (heating) values of each material before and after treatment were mentioned as having been recorded but were not reported for every material tested. Treated faeces and faecal simulant exhibited similar higher heating values[15] (HHV); HHVs for real faeces were in the range of 23.5 to 25.6 whilst for the faecal simulant they were in the range of 22.3 to 27.9. The values were primarily dependent on the temperature of the hydrothermal carbonisation process (Afolabi et al., 2014).

Ward *et al* (2014) considered the treatment of human waste by pyrolysis. The Sol-Char toilet, built by the University of Colorado Boulder, was developed to use concentrated sunlight to dry and pyrolyse human faeces in-situ thus limiting human exposure to untreated FS. In order to investigate the energy content of the chars produced following pyrolysis, Ward et al (2014) used both synthetic faeces – simulant recipe two from Table C-4 – and real faeces. Elemental analysis and oxygen bomb calorimetry was carried out on all char samples, the results of which are shown in Table C-7. No elemental analysis was carried out on raw feedstocks prior to pyrolysis.

[15] Higher heating value may be defined as "the amount of heat released by the unit mass or volume of fuel (initially at a temperature of 25°C) once it is combusted and the products have returned to a temperature of 25°C" (Basu, 2010). The higher heating value includes the latent heat of vaporisation.

Table C-7. Char yield, elemental composition and higher heating value (HHV) of human faecal and synthetic faecal chars where HF is human faeces, SF is synthetic faeces, C is carbon, H is hydrogen, N is nitrogen, S is sulphur and O is oxygen (Ward et al., 2014)

Char Type		%Char yield	%Ash	%C	%H	%N	%S	%O	HHV (MJ kg^{-1})
Feed-stock	Pyrolysis Temp (°C)								
HF	300	49.00	20.00	58.23	6.10	5.19	0.43[a]	10.05	25.57 ± 0.08
HF	450	29.00	37.10	50.67	1.90	4.76	0.43[a]	5.14	17.97 ± 0.32
HF	750	30.00	50.00	42.03	0.44	2.44	0.43[a]	4.69	13.83 ± 0.48
SF	300	39.00	8.30	64.05	7.48	3.10	0.05[b]	17.01	29.53 ± 0.91
SF	450	19.00	20.00	57.67	1.91	5.25	0.05	15.12	-
SF	750	14.00	38.30	56.17	0.38	2.72	0.05	2.38	18.92 ± 1.30

[a] Ward *et al* (2014) estimated the percentage of sulphur in human faeces char from the sulphur content of human faeces and adjusting for the decrease in sulphur content due to pyrolysis.
[b] Ward *et al* (2014) estimated the percentage of sulphur in synthetic faeces char from the sulphur content of synthetic faeces and adjusted for the decrease in sulphur content due to pyrolysis.

As shown in Table C-7 the char yield for synthetic faeces was lower at each pyrolysis temperature than for real faeces. Ward et al (2014) noted that the carbon content for synthetic chars decreased from 64% to 56% whilst for real faecal char it decreased from 58% to 42%. Hydrogen content was found to be similar for both chars and decreased with increasing pyrolysis temperature, as did oxygen content. Nitrogen content in human faecal char decreased with increasing pyrolysis temperature but didn't follow the same trend for synthetic char (Ward et al., 2014). Synthetic char had a HHV that was systematically higher than real faecal char by roughly 4 to 5 MJ kg^{-1} at each temperature tested (Ward et al., 2014). From these results Ward *et al* (2014) concluded that Wignarajah *et al* (2006) simulant two was not suitable as a model for the energy content of human faecal char. It was assumed that the difference in HHV was due to the difference in proportion of ash of the chars caused by the difference in inorganics content. Real human faeces have been found to contain between 10.0% and 15.0% elemental ash (inorganics) prior to pyrolysis as shown in Table C-7; however, the proportion of inorganics for simulant two from Table C-4 is only 5.0%. This is likely to contribute to the low inorganic mineral composition of the resulting simulant faeces char (Ward et al., 2014).

Serio *et al* (2016) used the faecal simulant developed by Wignarajah *et al* (2008) – without yeast – to test the efficacy of a torrefaction (mild pyrolysis operating at temperatures of less

than 300°C) reactor system for human faecal waste and related solid waste streams in space. The researchers also used canine faeces; however, there are "major differences in urinary metabolic pathways that would result in differences in chemical composition of faeces of dogs and humans" (Wignarajah et al., 2006). This would imply that canine faeces are not a suitable simulant for human faeces although canine faeces and dog food have also been used in studies investigating the biological composting of waste as noted by Wignarajah *et al* (2006). Dog food has been found to have a much higher protein content (21%) than real human faeces (Zuma, 2013). Table C-8 shows the elemental composition on a dry ash free (daf) basis of human faeces, dog faeces and the faecal simulant as reported by Serio *et al* (2016).

Table C-8. Comparison of elemental compositions (daf basis) of waste samples where db is dry basis, C is carbon, H is hydrogen, N is nitrogen, S is sulphur and O is oxygen (Serio et al., 2016)

Sample	**%Ash db**	**%C**	**%H**	**%N**	**%S**	**%O**
Human faeces	12.5	49.1	7.2	2.7	-	-
Canine faeces	22.1	49.1	7.4	4.1	0.6	38.7
Faeces simulant	13.4	57.4	9.0	1.1	0.1	32.5

As shown in Table C-8 the simulant faeces have a slightly higher ash content than real human faeces and canine faeces has a much higher percentage ash content, this would imply a higher inorganics content. The carbon content of the simulant is much higher than for real faeces which has the same carbon content as that reported for canine faeces. The nitrogen content in canine faeces is higher than that for human faeces. The elemental composition of human faeces differs to that reported by Ward *et al* (2014) as shown in Table C-7 likely due to temperature of analysis. Similar results for human faeces were reported by Fisher *et al* (1998) as shown in Table C-6 and for primary wastewater sludge reported by Danso-Boateng et al (2013) as shown in Table C-5. Slight differences in the values reported are likely due to normal variation of human faeces and differences in the elemental composition in the simulants reported in Table C-7 and Table C-8 are likely due to slight differences in the recipes used as well as temperature of analysis. The simulant was used to study the thermal characteristics of the system, for example, the temperature gradients and heating rate as it was assumed to behave in a similar manner to real human faeces. Canine faeces was used to study "the effects of temperature on the solid, liquid and gas composition, and the char properties in terms of hydrophobicity, bioactivity and odour" (Serio et al., 2016). It was assumed that, based on chemical composition, both the simulant and canine faeces would behave in a manner similar to human faeces if subjected to the torrefaction process.

To investigate the effect of ammonia concentration on biogas yield during anaerobic digestion Colón *et al* (2015) developed a simulant for excreta i.e. urine and faeces. The recipe

for the simulant was based on the faecal simulants developed by Wignarajah *et al* (2006) and the synthetic urine recipe developed by Putnam (1971). The chemical composition of the recipes is shown in Table C-9 Baker's yeast was used to represent the microbial content, microcrystalline cellulose and psyllium were used to represent carbohydrate/fibre proportion, oleic acid was used for fats and the miso paste was used to adjust nitrogen content and other chemical properties (Colón et al., 2015). Two urine concentrations were used to cover the normal range of nitrogen generally found in human faeces as the interest of the study was the effect of ammonia concentration since high ammonia concentrations have been found to inhibit anaerobic digestion.

Table C-9. Chemical composition of simulated faeces and urine; stages 3 and 4 refer to different phases during experimentation where NaCl is sodium chloride, KCl is potassium chloride, $CaCl_e$ is calcium chloride, $KHSO_4$ is potassium hydrogen sulfate, $MgSO_4$ is magnesium sulfate, KH_2PO_4 is potassium dihydrogen phosphate and $KHCO_3$ is potassium bicarbonate (Colón et al., 2015)

Faecal Simulant		**Urine Simulant**		
Component	**Amount (g L^{-1})**	**Component**	**Amount (g L^{-1})**	
			Stage 3	**Stage 4**
Water	800.00	Urea	9.30	14.20
Baker's yeast (dry)	60.00	Creatine	2.00	3.00
Microcrystalline cellulose	20.00	Ammonium citrate	1.00	2.00
Psyllium	35.00	NaCl	8.00	8.00
Miso paste	35.00	KCl	1.65	1.65
Oleic acid	40.00	$KHSO_4$	0.50	0.50
NaCl	4.00	$MgSO_4$	0.20	0.20
KCl	4.00	KH_2PO_4	1.75	1.75
$CaCl_2$	2.00	$KHCO_3$	0.50	0.50

The modifications made to the recipes by Colón *et al* (2015) were intended to ensure the simulant excreta mimicked the water holding capacity, consistency and chemical composition of real excreta. The simulant was noted to have the consistency of a thick paste but no attempt was made to match the rheological properties of real faeces as it was considered irrelevant to the study (Colón et al., 2015). Table C-10 shows the physicochemical properties of the simulant faeces and urine alongside published data from the characterisation of real faeces and urine.

Table C-10. Physicochemical properties of simulated faeces and simulated urine and comparison with data obtained from real samples where TS (%) is total solids, VS (%TS) is volatile solids, COD_t (g_{COD} g_{TS}^{-1}) it total chemical oxygen demand, COD_{SS} (g_{COD} g_{TS}^{-1}) is suspended chemical oxygen demand, COD_{dis} (g_{COD} g_{TS}^{-1}) is dissolved chemical oxygen demand, N_{tot} is total nitrogen, $N\text{-}NH_3$ is ammonia nitrogen and C/N is the carbon to nitrogen ratio (Colón et al., 2015)

	Faeces		Urine		
Property	**Simulant**	**Real faeces**	**Simulant (stage 3)**	**Simulant (stage 4)**	**Real urine**
Moisture (%)	81.6	65.0 – 85.0	97.6	96.5	95.0 – 98.0
TS (%)	18.4	15.0 – 35.0	2.4	3.5	2.5 – 3.7
VS (%TS)	88.5	80.0 – 92.0	49.5	62.5	60.0 – 75.0
COD_t:VS	1.51	1.56	-	-	-
COD_t (g_{COD} g_{TS}^{-1})	1.33	1.24	2.9	3.9	3.8 – 8.2
COD_{SS} (g_{COD} g_{TS}^{-1})	0.85	-	0.0	0.0	-
COD_{dis} (g_{COD} g_{TS}^{-1})	0.38	-	2.9	3.9	-
N_{tot} (% dry matter)	2.75	2.0 – 3.0	5200.0	8040.0	5000.0 – 8000.0
$N\text{-}NH_3$ (% N_{tot})	3.0	< 7.0	197.0	403.0	< 100.0
C/N	17.3	5.0 – 16.0	0.58	0.59	0.80
pH	5.3	4.6 – 8.4	6.0	6.0	6.0 – 8.2
Conductivity (mS cm^{-1})	5.7	-	23.0	24.0	16.0 – 22.0

For all of the properties tested both the faecal simulant and urine simulants were within the ranges of real faeces and urine. The moisture content and volatile solids content of the simulant faeces were on the higher side of the range for real faeces, whilst total solids content was on the lower end of the range. Total chemical oxygen demand was similar between the simulant and real faeces. Total nitrogen and ammonia nitrogen were very similar for both simulants and real faeces and urine. The carbon to nitrogen ratio is important to biodegradation processes. The ratio for the simulant faeces was found to be slightly higher than for real faeces and for simulant urine it was lower than real urine. The pH for all simulants was within the range for real faeces and urine. The conductivity reported for the simulant urine at both stages was higher than that reported for real urine. Colón *et al* (2015) found that the simulant excreta performed in a similar manner to real excreta when introduced into an anaerobic digestor and therefore provided a useful analogue for real human excreta.

Miller *et al* (2015) used the same recipe for synthetic faeces as Colón *et al* (2015) to investigate the destruction of the simulant in a bench-scale continuous supercritical water oxidation (SCWO) reactor unit. Table C-11 shows a comparison of the simulant faeces and real faeces as reported by Miller *et al* (2015). The results reported are the same as those reported by Colón *et al* (2015). There is slight variation in the elemental composition of the synthetic

faeces compared to the results reported by Serio *et al* (2016) in Table C-11 which may be due to the addition of the yeast which was not included in the simulant used by Serio *et al* (2016). Due to the similarities in the physicochemical properties tested between the simulant faeces and real human faeces it was assumed that the simulant would behave in a similar manner to real faeces. The results of the experiments served to define the optimal operating conditions for process variables that would later be used to design and operate a larger scale SCWO unit (Miller et al., 2015).

Table C-11. Physicochemical properties of simulant faeces compared with data obtained from real samples where TS (%) is total solids, VS (%TS) is volatile solids, COD_t ($g_{COD}\ g_{TS}^{-1}$) it total chemical oxygen demand, COD_{SS} ($g_{COD}\ g_{TS}^{-1}$) is suspended chemical oxygen demand, COD_{dis} ($g_{COD}\ g_{TS}^{-1}$) is dissolved chemical oxygen demand, N_{tot} is total nitrogen, $N\text{-}NH_3$ is ammonia nitrogen and C/N is the carbon to nitrogen ratio, C is carbon, H is hydrogen, N is nitrogen and S is sulpher (Miller et al., 2015)

Property	Simulant	Real faeces
Moisture (%)	81.6	65.0 – 85.0
TS (%)	18.4	15.0 – 35.0
VS (%TS)	88.5	80.0 – 92.0
COD_t:VS	1.51	1.56
COD_t ($g_{COD}\ g_{TS}^{-1}$)	1.33	1.24
N_{tot} (% dry matter)	2.75	2.0 – 3.0
$N\text{-}NH_3$ (% N_{tot})	3.0	< 7.0
C/N	17.3	5.0 – 16.0
pH	5.3	4.6 – 8.4
Conductivity ($mS\ cm^{-1}$)	5.7	-
C (%)	44.9	-
H (%)	7.1	-
N (%)	2.1	-
S (%)	0.2	-

Podichetty *et al* (2014) considered a simulant made of salt, rice, soy bean paste (miso) and water as well as pig and chicken faeces, wheat flour and red potatoes in order to evaluate the application of viscous heating to treat faecal matter. Structural, thermal and viscoelastic properties of the simulants compared to real faeces was of interest in this case. Heat was generated in the simulants by the application of shear stress with an extruder; the heat generated was considered sufficient to sanitise faeces. Although it was found that wheat flour had the closest match to human faeces with change in viscosity over a range of shear rate, red potato was chosen as a suitable simulant when matched with moisture content for practical experiments and computational fluid dynamics (CFD) modelling. Table C-12 shows the properties of the red potato used for CFD modelling alongside data reported from the literature for real human faeces. The density of the red potato and real faeces were comparable as shown in Table C-12. The thermal properties reported by Velkushanova et al

(2017) were estimated at approximately 20.0°C but the temperature at which the thermal properties of the red potato was determined was not reported by Podichetty *et al* (2014). Therefore, it was not possible to compare the thermal properties of red potato with that of faeces.

Table C-12. Properties of red potato used for CFD modelling (Podichetty et al., 2014)

Property	**Red potato**	**Faeces**
Density (kg m^{-3})	1030.00	1060.00 – 1090.00[a]
Thermal conductivity (W m^{-1} K^{-1})	0.03	0.39[b]
Heat capacity (J kg^{-1} K^{-1})	4186.00	2150.49[b]

[a] Penn et al (2018)
[b] Value reported for faeces from urine diversion toilet (Velkushanova et al., 2017)

Other simulants which were noted in the literature include chicken litter, monkey faeces and refried beans (Wignarajah et al., 2006). According to Wignarajah *et al* (2006) none are a suitable replacement for human faeces given the significant differences in terms of both chemical and physical properties. Chicken litter consists of bedding straw and chicken faeces which, when compared with human faeces, has a higher nitrogen content, lower water and microbial content and a much higher mineral content (Wignarajah et al., 2006). Refried beans were also used to simulate human faeces although analysis showed that they were much higher in protein content than human faeces. Yeo *et al* (1998) found mashed potatoes, brownie mix, peanut butter, pumpkin pie filling and some commercially available simulants such as FECLONE® had been previously used as faecal simulants. However, Yeo *et al* (1998) discounted them all given that they did not sufficiently bind water to themselves and so released water too quickly compared to real human faeces.

The simulants discussed in this appendix have been found to have been largely based on the work done by Wignarajah *et al* (2006). There appears to be a distinction between those simulants developed to mimic the physical properties e.g. rheological properties, and those developed to mimic the chemical composition and water-holding capacity of real faeces

D Analysis and Characterisation of Real Faecal Sludge

Appendix D provides a comprehensive overview of the literature found regarding the analysis and characterisation of real faecal sludge. A list of references alongside properties analysed, source of the sludge analysed or the containment technology, the methodology used for analysis and any comments regarding the reported data. The frequency of each characterisation test was also determined.

The following symbols are used in the table of properties analysed and methods:

- x Result reported
- o No result provided but property mentioned
- - Results reported from other sources
- N/A Not applicable
- N/S Not specified
- U Unknown

Note that where rows contain red font it is indicative of results which were reported in other papers.

Real faecal sludge characterisation tests: Methodologies and properties analysed

| | | Physical and Aggregate Properties | | |

E Statistical Results for Faecal Sludge Simulant Characterisation

Appendix E provides the interquartile range, median and average values for the simulants tested as part of this work as well as for the real faecal sludge used for comparison. Values are reported for the following tests: moisture content; total solids; volatile solids; the ratio of volatile solids to total solids; ash; pH; density; thermal conductivity and dry and wet specific heat capacity.

Table E-1. Water content (w (%)) and total solids (TS (%)) statistical results for simulants and for real faecal sludge from a variety of onsite containment technologies (OCT) and sources where n is the total number of samples tested, Q2 is the median, IQR is interquartile range, $\bar{x}$ is the mean, UD is urine diversion toilet, Sch. Toilets is school toilets, Cml. ST is commercial septic tank, 2CP (Ser) is two cesspools in series, 1CP is one cesspool, ST is septic tank, Fl is fluidised, bott is bottom and blank cells are indicative of no result being reported

	Simulant or Type of OCT	n	w (%)			n	TS (%)		
			Q2	IQR	$\bar{x}$		Q2	IQR	$\bar{x}$
PRG (2014a; 2014b)	FF				76.00				24.00
	Sch. Toilets				62.69				37.27
	CAB				77.00				23.00
	UD				60.71				44.97
	Unimproved Pit				41.59				58.41
	Wet VIP				78.86				21.14
	Dry VIP				79.24				20.76
Koottatep *et al* (2012)	Cml. ST – Fl. Sludge	8	72.37	6.58	77.16	8	26.73	8.05	21.72
	Cml. ST – Bott. Sludge	8	75.82	8.05	80.53	8	23.44	7.03	19.00
	2CP (Ser) – Fl. Sludge	12	29.21	13.76	40.73	12	3.25	2.84	3.45
	2CP (Ser) – Bott. Sludge	10	29.72	8.43	31.64	10	0.79	1.11	1.10
	1CP – Fl. Sludge	30	95.97	6.49	94.04	31	4.06	7.41	6.09
	1CP – Bott. Sludge	15	99.12	0.77	98.88	16	0.92	0.70	1.01
	ST – Fl. Sludge	10	96.73	4.38	96.01	10	3.26	3.85	3.98
	St – Bott Sludge	10	98.94	1.48	98.22	10	0.95	1.53	1.74
	ND4	15	83.15	0.80	83.19	15	16.85	0.80	16.81
	ND3	15	85.45	1.12	85.25	15	14.55	1.12	14.75
	ND2	15	83.31	11.86	81.74	15	16.69	11.86	18.26
	ND1	15	86.88	10.27	85.60	15	13.12	10.27	14.40
	SE80	15	83.03	0.23	83.08	15	16.97	0.23	16.92
	BM6	15	85.70	0.85	85.75	15	14.30	0.85	14.25
	BM5	15	86.06	0.43	86.15	15	13.94	0.43	13.85
	BM4	15	80.22	1.37	80.20	15	19.78	1.37	19.80
	BM3	15	79.62	0.87	79.82	15	20.38	0.87	20.18
	BM2	15	80.97	0.82	81.06	15	19.03	0.82	18.94
	BM1	15	81.01	1.00	80.84	15	18.99	1.00	19.16
	SS9	15	80.63	1.06	80.49	15	19.37	1.06	19.51

Table E-2. Volatile solids (VS (%)), average VS to average TS (VS_{ave}:TS_{ave}) and ash (%) statistical results for simulants and for real faecal sludge from a variety of onsite containment technologies (OCT) and sources where n is the total number of samples tested, Q2 is the median, IQR is interquartile range, $\bar{x}$ is the mean, UD is urine diversion toilet, Sch. Toilets is school toilets, Cml. ST is commercial septic tank, 2CP (Ser) is two cesspools in series, 1CP is one cesspool, ST is septic tank, Fl is fluidised, bott is bottom and blank cells are indicative of no result being reported

	Simulant or Type of OCT	**n**	**VS (%)**			**VS_{ave}:TS_{ave}**	**n**	**Ash (%)**		
			Q2	**IQR**	**$\bar{x}$**			**Q2**	**IQR**	**$\bar{x}$**
PRG (2014a; 2014b)	FF				87.00	3.63				13.00
	Sch. Toilets				33.58	0.90				71.81
	CAB				49.00	2.13				51.00
	UD				41.95	0.93				43.96
	Unimproved Pit				28.19	0.48				71.81
	Wet VIP				53.61	2.54				46.39
	Dry VIP				57.81	2.78				42.19
Koottatep *et al* (2012)	Cml. ST – Fl. Sludge	8	2.26	1.24	2.52	0.12	8	24.73	7.56	19.19
	Cml. ST – Bott. Sludge	8	0.68	0.49	1.06	0.06	8	22.73	6.83	18.13
	2CP (Ser) – Fl. Sludge	12	1.98	1.78	5.19	1.50	12	1.09	0.91	1.17
	2CP (Ser) – Bott. Sludge	10	0.59	0.65	0.72	0.66	10	0.34	0.29	0.45
	1CP – Fl. Sludge	31	3.10	2.49	3.59	0.59	31	1.46	3.22	2.87
	1CP – Bott. Sludge	15	0.68	0.54	0.72	0.72	15	0.21	0.20	0.28
	ST – Fl. Sludge	10	2.37	2.16	2.60	0.65	10	0.75	1.80	1.38
	St – Bott Sludge	10	0.75	0.92	1.23	0.70	10	0.19	0.60	0.52
	ND4	6	76.32	0.15	76.25	4.54	6	23.68	0.15	23.75
	ND3	6	76.54	0.16	76.24	4.79	6	23.46	0.16	23.76
	ND2	6	78.02	19.96	67.27	3.68	6	20.14	1.91	19.76
	ND1	6	73.63	3.92	72.24	4.31	6	26.37	3.92	27.76
	SE80	6	91.22	0.17	90.74	5.17	6	8.78	0.17	9.26
	BM6	6	80.13	1.01	80.00	5.61	6	19.87	1.01	20.00
	BM5	6	79.78	0.83	79.71	5.64	6	20.22	0.83	20.29
	BM4	6	84.63	0.29	84.65	4.24	6	15.37	0.29	15.35
	BM3	6	81.47	5.63	81.67	4.10	6	18.53	5.63	18.33
	BM2	6	83.01	0.23	83.13	4.38	6	16.99	0.34	16.92
	BM1	6	83.48	2.73	83.29	4.39	6	16.27	2.39	16.52
	SS9	6	83.09	0.20	82.54	4.23	6	16.91	0.20	17.46

Table E-3. pH and density (ρ (kg m^{-3})) statistical results for simulants and for real faecal sludge from a variety of onsite containment technologies (OCT) and sources where n is the total number of samples tested, Q2 is the median, IQR is interquartile range, $\bar{x}$ is the mean, UD is urine diversion toilet, Sch. Toilets is school toilets, Cml. ST is commercial septic tank, 2CP (Ser) is two cesspools in series, 1CP is one cesspool, ST is septic tank, Fl is fluidised, bott is bottom and blank cells are indicative of no result being reported

	Simulant or Type of OCT	**n**	**pH**			**n**	**ρ (kg m^{-3})**		
			Q2	**IQR**	**$\bar{x}$**		**Q2**	**IQR**	**$\bar{x}$**
PRG (2014a; 2014b)	FF								
	Sch. Toilets				8.24				1405.55
	CAB				7.44				1350.10
	UD				7.23				1450.37
	Unimproved Pit				6.65				921.45
	Wet VIP				7.59				1447.78
	Dry VIP				7.60				1379.72
Koottatep *et al* (2012)	Cml. ST						1105.00	20.00	1100.00
	2CP (Ser)						1120.00	25.00	1126.00
	1CP						1120.00	17.50	1121.11
	ST						1110.00	15.00	1108.57
	ND4	6	7.00	0.01	6.99	6	1490.45	3.06	1492.18
	ND3	6	6.40	0.02	6.39	6	1397.63	74.42	1433.21
	ND2	6	5.66	0.00	5.66	6	1295.37	33.70	1280.26
	ND1	6	5.30	0.01	5.29	6	1435.32	24.01	1445.85
	SE80	6	5.30	0.00	5.30	6	1308.06	4.43	1308.24
	BM6	6	5.68	0.02	5.71	6	1375.83	4.19	1374.17
	BM5	6	5.72	0.11	5.70	6	1357.79	12.92	1356.87
	BM4	6	5.13	0.01	5.12	6	1369.71	2.25	1369.61
	BM3	6	5.28	0.02	5.30	9	687.52	680.91	800.92
	BM2	6	5.26	0.04	5.26	6	1353.14	7.26	1351.19
	BM1	6	5.27	0.01	5.26	6	1342.97	9.47	1342.70
	SS9	6	5.21	0.05	5.21	6	1341.69	4.43	1344.06

Table E-4. Thermal conductivity (λ (W $m^{-1}K^{-1}$) statistical results for simulants and for real faecal sludge from a variety of onsite containment technologies (OCT) and sources where n is the total number of samples tested, Q2 is the median, IQR is interquartile range, $\bar{x}$ is the mean, UD is urine diversion toilet, Sch. Toilets is school toilets, Cml. ST is commercial septic tank, 2CP (Ser) is two cesspools in series, 1CP is one cesspool, ST is septic tank, Fl is fluidised, bott is bottom and blank cells are indicative of no result being reportedd

	Simulant or Type of OCT	n	λ (W $m^{-1}K^{-1}$)		
			Q2	IQR	$\bar{x}$
PRG (2014a; 2014b)	FF				
	Sch. Toilets				0.55
	CAB				0.60
	UD				0.39
	Unimproved Pit				0.29
	Wet VIP				0.55
	Dry VIP				0.54
Koottatep *et al* (2012)	Cml. ST – Mixed sample from Nakhon (dried sample)				0.03
	2CP (Ser) – Mixed sample from Lampang (dried sample)				0.04
	1CP – Mixed sample from Ratchaburi (dried sample)				0.05
	ST – Mixed sample from Nonthaburi (67% DS)				0.14
	ST – Mixed sample from Nonthaburi (dried sample)				0.04
	ND4	6	0.0687	0.0026	0.0683
	ND3	6	0.0700	0.0044	0.0701
	ND2	6	0.0582	0.0024	0.0584
	ND1	6	0.0607	0.0045	0.0614
	SE80	6	0.0585	0.0003	0.0585
	BM6	6	0.0629	0.0003	0.0626
	BM5	6	0.0589	0.0012	0.0587
	BM4	6	0.0510	0.0045	0.0516
	BM3	6	0.0526	0.0024	0.0524
	BM2	6	0.0581	0.0004	0.0580
	BM1	6	0.0583	0.0012	0.0520
	SS9	6	0.0463	0.0002	0.0464

Table E-5. Dry specific heat capacity (c_{dry} (J kg^{-1}K^{-1})) and wet specific heat capacity (c_{wet} (J kg^{-1}K^{-1})) statistical results for simulants and for real faecal sludge from a variety of onsite containment technologies (OCT) and sources where n is the total number of samples tested, Q2 is the median, IQR is interquartile range, $\bar{x}$ is the mean, UD is urine diversion toilet, Sch. Toilets is school toilets, Cml. ST is commercial septic tank, 2CP (Ser) is two cesspools in series, ICP is one cesspool, ST is septic tank, Fl is fluidised, bott is bottom and blank cells are indicative of no result being reported

	Simulant or Type of OCT	**n**	**°C**	**c_{dry} (J kg^{-1}K^{-1})**			**c_{wet} (J kg^{-1}K^{-1})**		
				Q2	**IQR**	**$\bar{x}$**	**Q2**	**IQR**	**$\bar{x}$**
PRG (2014a; 2014b)	FF								
	Sch. Toilets		20.0						3509.58
	CAB		20.0						3268.45
	UD		20.0						2199.58
	Unimproved Pit		20.0						2366.89
	Wet VIP		20.0						2422.33
	Dry VIP		20.0						2530.54
Koottatep *et al* (2012)	Cml. ST – Nakhon Ratchasima		25.0			1330.00			
	2CP (Ser) – Mixed sampled from Nonthaburi		25.0			1870.00			
	1CP – Mixed sampled from Ratchaburi		25.0			1390.00			
	ST – Mixed sample from Lampang		25.0			1570.00			
	ST – Nonthanburi		25.0			1670.00			
	ND4	4	26.8	1240.13	49.60	1264.44	2555.67	27.40	2569.10
	ND3	4	26.8	1235.68	32.67	1238.16	2553.22	18.05	2554.59
	ND2	4	26.8	1488.23	38.96	1497.72	2722.55	21.09	2727.68
	ND1	4	26.8	1203.71	26.55	1204.90	2548.80	14.55	2549.45
	SE80	4	26.8	1448.60	33.20	1459.17	2682.93	18.20	2688.73
	BM6	4	26.8	1561.40	213.25	1721.17	2770.25	114.81	2856.27
	BM5	4	26.8	1633.55	101.91	1601.50	2809.91	54.83	2792.67
	BM4	4	26.8	1389.50	775.17	1402.21	2629.98	430.58	2637.04
	BM3	4	26.8	1355.60	75.78	1344.82	2615.32	41.98	2609.35
	BM2	4	26.8	1492.56	43.88	1504.00	2695.23	24.24	2701.54
	BM1	4	26.8	1505.96	135.30	1543.59	2702.79	74.74	2723.58
	SS9	4	26.8	1497.11	134.93	1502.94	2693.55	74.76	2696.78